Humberto Blanco
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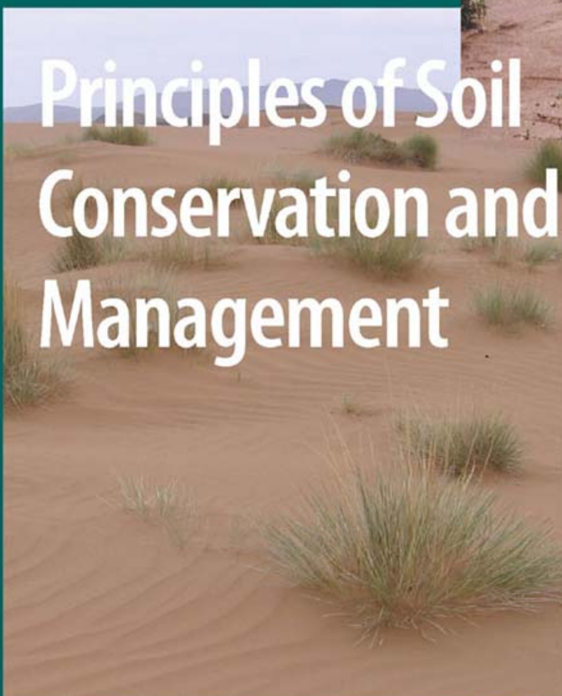
Principles of Soil Conservation and Management



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Preface

Management and conservation of soil and water resources are critical to human well-being. Their prudent use and management are more important now than ever before to meet the high demands for food production and satisfy the needs of an increasing world population. Despite the extensive research and abundant literature on soil and water conservation strategies, concerns of worldwide soil degradation and environmental pollution remain high. Several of the existing textbooks deal with principles of soil erosion, measurement, and modeling of soil erosion, and climatic (rainfall and wind) factors affecting the rate and magnitude of erosion. Yet, a state-of-the-science textbook for graduate and undergraduate students with emphasis on soil management to address the serious problems of soil erosion and the attendant environmental pollution is needed. Managing soils under intensive use and restoring eroded/degraded soils are top priorities to a sustained agronomic and forestry production while conserving soil and water resources. Management must come before conservation for the restoration and improvement of vast areas of world's eroded and degraded soils and ecosystems.

Thus, this textbook presents a comprehensive review and discussion of the: (1) severity and implications of soil erosion, (2) principles of management and conservation of soil and water resources, (3) impacts of water, wind and tillage erosion on soil resilience, carbon (C) sequestration and dynamics, CO_2 emissions, and food security, and (4) risks of soil erosion and the attendant relationships with the projected climate change and vice versa. It differs from other textbooks in that it incorporates detailed discussions about biological/agronomic management practices (e.g., no-till systems, organic farming, agroforestry, buffer strips, and crop residues), tillage erosion, C dynamics and sequestration, non-point source pollution (e.g. hypoxia), soil quality and resilience, and the projected global climate change.

This textbook specifically links the soil and water conservation issues with the restorative practices, soil resilience, C sequestration under different land use and soil management systems, projected global climate change, and global food security. This textbook also synthesizes current information on a new paradigm of soil management which is soil quality. Being a textbook of global relevance, it links and applies the leading research done in developed countries such as in the USA to contrasting scenarios of soil erosion problems in the developing countries.

Soil erosion history and the basic principles of water and wind erosion (e.g., factors, processes) have been widely discussed in several textbooks. Thus, the present volume presents only a condensed treatise on these topics. Major attention is given to management rather than to generic factors and processes of erosion. Chapter 1 reviews the implications of soil erosion in the USA and the global hotspots and presents the state-of-knowledge of soil and water conservation research and practices. Chapter 2 synthesizes the processes and factors of water erosion, whereas Chapter 3 reviews the factors and processes of wind erosion with emphasis on the management and control. Chapter 4 discusses the water and wind erosion models and presents examples of calculations of runoff and soil erosion rates. Chapter 5 introduces a relatively new topic in soil and water conservation research, which is tillage erosion. Discussions on tillage erosion have been practically ignored in soil conservation textbooks. Yet, it is an essential topic provided that erosion by tillage can be equal to or even higher than that by water or wind, especially in rolling agricultural landscapes.

A larger portion of this textbook from Chapters 6 to 11 is devoted to the management and control of soil erosion. These six Chapters provide comprehensive and thorough assessment of integrated management techniques and approaches to manage and conserve soil and water resources for diverse land uses. Benefits of crop residues, conservation buffers, agroforestry systems, crop rotations, and conservation tillage (e.g., no-till) systems are discussed. Chapter 11 reviews the different types of mechanical structures used for erosion control. Erosion in forestlands, rangelands, and pasturelands is discussed in Chapters 12 and 13. Chapter 14 covers the current topics addressing the implications of soil erosion and water runoff to nutrient/chemical transport causing eutrophication and hypoxia or ‘dead zones” in coastal ecosystems around the world. Water pollution caused by the excessive and indiscriminate use of agricultural chemicals on agricultural, forestry, and urban lands is discussed.

Chapter 15 describes management strategies for restoring eroded, compacted, saline and sodic, acidic, and mined soils, whereas inherent potential of the intensively managed, degraded, and misused soils to recover from the degradation forces is discussed in Chapter 16. Chapter 17 introduces a new topic in soil management and conservation concerning sequestration of C in terrestrial ecosystems and net emissions of CO_2 to the atmosphere. This chapter also discusses the transfers of soil C with sediment and runoff water and its fate. Towards the end of the textbook, relations of soil management with soil quality, food security, and global climate change are described (Chapters 18, 19, and 20). These chapters uniquely address the impacts of projected global warming on soil erosion risks and the attendant decline in food production. Finally, Chapter 21 addresses trends in soil conservation and management research as well as research needs for an effective soil and water conservation and management. It identifies possible shortcomings of past and current research work in soil and water conservation and suggests measures for improvement.

This textbook is suitable for undergraduate and graduate students in soil science, agronomy, agricultural engineering, hydrology, and management of natural resources and agricultural ecosystems. It is also of interest to soil conservationists and policymakers to facilitate understanding of principles of soil erosion and implementing strategic measures of soil conservation and management. The contents of this textbook are easily comprehended by students with a basic knowledge of introductory soils, hydrology, and climatology. Students will gain a better understanding of the basic concepts by following solved problems and doing additional problems given at the end of each chapter. The select problems are designed to further enhance the understanding of the material discussed in each chapter. Application of basic concepts is depicted by pictures from diverse management systems, soils, and ecoregions.

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Contents

Chapter 1
Soil and Water Conservation

1.1 Why Conserve Soil?

Soil is the most fundamental and basic resource. Although erroneously dubbed as "dirt" or perceived as something of insignificant value, humans can not survive without soil because it is the basis of all terrestrial life. Soil is a vital resource that provides food, feed, fuel, and fiber. It underpins food security and environmental quality, both essential to human existence. Essentiality of soil to human well-being is often not realized until the production of food drops or is jeopardized when the soil is severely eroded or degraded to the level that it loses its inherent resilience (Fig. 1.1).

Traditionally, the soil's main function has been as a medium for plant growth. Now, along with the increasing concerns of food security, soil has multi-functionality including environmental quality, the global climate change, and repository for ur-

Fig. 1.1 Soil erosion not only reduces soil fertility, crop production, and biodiversity but also alters water quality and increases risks of global climate change and food insecurity (Courtesy USDA-NRCS)

Table 1.1 Multifunctionality of soils

Food security, biodiversity, and urbanization	Water quality	Projected global climate change	Production of biofuel feedstocks
• Food • Fiber • Housing • Recreation • Infrastructure • Waste disposal • Microbial diversity • Preservation of flora and fauna	• Filtration of pollutants • Purification of water • Retention of sediment and chemicals • Buffering and transformation of chemicals	• Sink of CO_2 and CH_4 • C sequestration in soil and biota • Reduction of nitrification • Deposition and burial of C-enriched sediment	• Bioenergy crops (e.g., warm season grasses and short-rotation woody crops) • Prairie grasses

ban/industrial waste. World soils are now managed to: (1) meet the ever increasing food demand, (2) filter air, (3) purify water, and (3) store carbon (C) to offset the anthropogenic emissions of CO_2 (Table 1.1).

Soil is a non-renewable resource over the human time scale. It is dynamic and prone to rapid degradation with land misuse. Productive lands are finite and represent only $<$11% of earth's land area but supply food to more than six billion people increasing at the rate of 1.3% per year (Eswaran et al., 2001). Thus, widespread degradation of the finite soil resources can severely jeopardize global food security and also threaten quality of the environment. Conserving soil has many agronomic, environmental, and economical benefits. The on- and off-site estimated costs of erosion for replenishing lost nutrients, dredging or cleaning up water reservoirs and conveyances, and preventing erosion are very high and estimated at US$ 38 billion in the USA and about US$ 400 billion in the world annually (Uri, 2000; Pimentel et al., 1995). In the USA, the estimated cost of water erosion ranges from US$ 12 to US$ 42 billion while that of wind erosion ranges from US$ 11 to US$ 32 billion (Uri, 2000).

The need to maintain and enhance multi-functionality necessitates improved and prudent management of soil for meeting the needs of present and future generations. The extent to which soil stewardship and protection is professed determines the sustainability of land use, adequacy of food supply, the quality of air and water resources, and the survival of humankind. Soil conservation has been traditionally discussed in relation to keeping the soil in place for crop production. Now, soil conservation is evaluated in terms of its benefits to increasing crop yields, reducing water pollution, and mitigating concentration of greenhouse gases in the atmosphere.

1.2 Agents that Degrade Soil

Water and wind erosion are two main agents that degrade soils. Water erosion affects nearly 1,100 million hectares (Mha) worldwide, representing about 56% of the total degraded land while wind erosion affects about 28% of the total degraded land area (Oldeman, 1994). Runoff washes away the soil particles from sloping and bare lands while wind blows away loose and detached soil particles from flat and unprotected lands. Another important pathway of soil redistribution, often overlooked, is the tillage erosion caused by plowing, which gradually moves soil downslope in plowed fields with adverse on-site effects on crop production. Soil compaction, poor drainage, acidification, alkalinization and salinization are other processes that also degrade soils in specific conditions of parent material, climate, terrain, and water management.

1.3 Soil Erosion

There are two main types of erosion: *geologic and accelerated erosion. Geologic erosion* is a normal process of weathering that generally occurs at low rates in all soils as part of the natural soil-forming processes. It occurs over long geologic time horizons and is not influenced by human activity. The wearing away of rocks and formation of soil profiles are processes affected by the slow but continuous geologic erosion. Indeed, low rates of erosion are essential to the formation of soil. In contrast, soil erosion becomes a major concern when the rate of erosion exceeds a certain threshold level and becomes rapid, known as *accelerated erosion*. This type of erosion is triggered by anthropogenic causes such as deforestation, slash-and-burn agriculture, intensive plowing, intensive and uncontrolled grazing, and biomass burning.

Control and management of soil erosion are important because when the fertile topsoil is eroded away the remaining soil is less productive with the same level of input. While soil erosion can not be completely curtailed, excessive erosion must be reduced to manageable or tolerable level to minimize adverse effects on productivity. Magnitude and the impacts of soil erosion on productivity depend on soil profile and horizonation, terrain, soil management, and climate characteristics. The estimated average tolerance (T) level of soil erosion used in soil and water conservation planning in the USA is 11 $Mg\,ha^{-1}\,yr^{-1}$. The T value is the amount of soil erosion that does not significantly decrease soil productivity. The specific rates of maximum tolerable limits of erosion vary with soil type. In fact, moderate soil erosion may not adversely affect productivity in well-developed and deep soils, but the same amount of erosion may have drastic effects on shallow and sloping soils. Thus, critical limits of erosion must be determined for each soil, ecoregion, land use, and the farming system.

1.3.1 Water Erosion

On a global scale, water erosion is the most severe type of soil erosion (Fig. 1.1). It occurs in the form of splash/interrill, rill, gully, tunnel, streambank, and coastal erosion. Different forms of erosion are discussed in detail in Chapter 2. Runoff occurs when precipitation rates exceed the water infiltration rates. Both raindrop impact and water runoff can cause soil detachment and transport. Unlike wind erosion, water erosion is a dominant form of erosion in humid, and sub-humid, regions characterized by frequent rainstorms. It is also a problem in arid and semiarid regions where the limited precipitation mostly occurs in the form of intense storms when the soil is bare and devoid of vegetal cover. One of the spectacular types of water erosion is the concentrated gully erosion which can cause severe soil erosion even in a single event of high rainfall intensity. Excessive gully erosion can wash out crops, expose plant roots, and lower ground water table while adversely affecting plant growth and landscape stability. Gullying is a major source of sediment and nutrient loss. It causes drastic alterations in landscape aesthetics and removes vast amounts of sediment.

Sedimentation at the lower end of the fields in depressional sites can bury crops, damage field borders, and pollute water bodies. Gullies dissect the field and exacerbate the non-point source pollution (e.g., sediment, chemicals) to nearby water sources. Gullies undercut and split croplands and alter landform features and watercourses. In the USA, soil erosion by gully erosion has been measured at 100 $Mg\,ha^{-1}yr^{-1}$ and represents about 21–275% of the interrill and rill erosion (USDA, 1996). In mountainous terrains and structurally fragile soils subjected to intense rains, total erosion from gullies can be as high as that from other types of erosion.

1.3.2 Wind Erosion

Wind erosion is a widespread phenomenon, especially in arid and semi-arid regions. It is a dominant geomorphic force that has reshaped the earth. Most of the material carried by wind consists of silt-sized particles. Deposition of this material, termed as “loess”, has developed into very fertile and deep soils. The thickness of most loess deposits ranges between 20 and 30 m, but it can be as thick as 335 m (e.g., Loess Plateau in China). Extensive deposits of loess exist in northeastern China, Midwestern USA, Las Pampas of Argentina, and central Europe.

Excessive wind erosion due to soil mismanagement has, however, caused the barren state of many arid lands (Fig. 1.2). Anthropogenic activities set the stage for severe wind erosion by directly influencing soil surface conditions through deforestation and excessive tillage. Wind erosion is prominent but not unique to arid regions. High winds, low precipitation ($\leq$300 mm annually), high evapotranspiration, reduced vegetative cover, and limited soil development are the main drivers of wind erosion in arid and semiarid regions. Rates of wind erosion increase in the order of: arid>semiarid> dry subhumid areas>humid areas. Unlike water, wind has the ability to move soil particles up- and down-slope and can pollute both air and water. While arid lands are more prone to wind erosion than humid ecosystems, any cultivated soil that is seasonally disturbed can be subject to eolian processes in windy environments.

Fig. 1.2 Wind erosion reduces vegetative cover and forms large sand dunes in arid regions (Photo by H. Blanco)

Wind erosion not only alters the properties and processes of the eroding soil but also adversely affects the neighboring soils and landscapes where the deposition may occur. Landscapes prone to wind erosion often exhibit an impressive network of wind ripples (<2 m high) (Fig. 1.2). Formation of sand dunes in deserts or along beaches is a sign of excessive wind erosion. Sand dunes can be as high as 200 m in desert regions of the world (e.g., Saudi Arabia). The smaller sand dunes often migrate and form larger sand dunes. There are fast moving as well as slow drifting dunes.

1.4 History of Soil Erosion

Accelerated erosion is as old as agriculture. It dates back to the old civilizations in Mesopotamia, Greece, Rome, and other regions in the Middle East (Bennett, 1939). The collapse of great ancient civilizations in Mesopotamia along the Tigris-Euphrates Rivers illustrates the consequences when lands are irreversibly degraded. Lessons from the past erosion and consequences for the demise of ancient civilizations have been amply cited and discussed in several textbooks. Indeed, Hugh Hammond Bennett, recognized as the "Father of Soil Conservation" in the U.S., described in his well-known textbook in detail the historical episodes and consequences of severe erosion (Bennett, 1939). Troeh et al. (2004) also reviewed past and current erosion rates around the world. Knowledge of the historic erosion is critical to understanding the severity and consequences of erosion and developing strategies for effective management of present and future soil erosion. Thus, readers are referred to other textbooks for details on historic rates of erosion. This textbook primarily focuses on the processes and strategies for effectively managing soil erosion.

1.5 Consequences of Soil Erosion

Accelerated soil erosion causes adverse agronomic, ecologic, environmental, and economic effects both on-site and off-site. Not only it affects agricultural lands but also quality of forest, pasture, and rangelands. Cropland soils are, however, more susceptible to erosion because these soils are often left bare or with little residue cover between the cropping seasons. Even during the growing season, row crops are susceptible to soil erosion. The on-site consequences involve primarily the reduction in soil productivity, while the off-site consequences are mostly due to the sediment and chemicals transported away from the source into natural waters by streams and depositional sites by wind.

1.5.1 On-site Problems

The primary on-site effect of erosion is the reduction of topsoil thickness, which results in soil structural degradation, soil compaction, nutrient depletion, loss of soil organic matter, poor seedling emergence, and reduced crop yields (Fig. 1.3). Removal of the nutrient-rich topsoil reduces soil fertility and decreases crop yield. Soil erosion reduces the functional capacity of soils to produce crops, filter pollutants, and store C and nutrients. One may argue that, according to the law of conservation of matter, soil losses by erosion in one place are compensated by the gains

Fig. 1.3 Runoff sediment pollutes nearby water sources (Courtesy USDA-NRCS)

at another place. The problem is that the eroded soil may be deposited in locations where either no crops can be grown or it buries and inundates the crops in valleys.

1.5.2 Off-site Problems

Water and wind erosion preferentially remove the soil layers where most agricultural chemicals (e.g., nutrients, pesticides) are concentrated. Thus, off-site transport of sediment and chemicals causes pollution, sedimentation, and silting of water resources (Fig. 1.3). Sediment transported off-site alters the landscape characteristics, reduces wildlife habitat, and causes economic loss. Erosion also decreases livestock production through reduction in animal weight and forage production, damages water reservoirs and protective shelterbelts, and increases tree mortality. Accumulation of eroded materials in alluvial plains causes flooding of downstream croplands and water reservoirs. Soil erosion also contributes to the projected global climate change. Large amounts of C are rapidly oxidized during erosion, exacerbating the release of CO_2 and CH_4 to the atmosphere (Lal, 2003).

Wind erosion causes dust pollution, which alters the atmospheric radiation, reduces visibility, and causes traffic accidents (Fig. 1.4). Dust particles penetrate into buildings, houses, gardens, and water reservoirs and deposit in fields, rivers, lakes, and wells, causing pollution and increasing maintenance costs. Dust storms transport fine inorganic and organic materials, which are distributed across the wind path. Most of the suspended particles are transported off-site and are deposited hundreds or even thousands of kilometers far from the source. Airborne fine particulate matter with diameters of 10 μm (PM10) and 2.5 μm (PM2.5) pose an increasing threat to human and animal health, industrial safety, and food processing plants. Finer particles float in air and are transported at longer distances than coarser particles. Particle size of the deposited eolic material decreases with increase in distance from

Fig. 1.4 Air pollution during the Dust Bowl (Courtesy USDA-NRCS)

Table 1.2 Some of the erosion-induced soil degradation processes

Physical Processes	Chemical Processes	Biological Processes
Increase in: • Surface sealing • Crusting • Compaction • Deflocculation • Sand content **Decrease in:** • Topsoil depth • Soil structural stability • Macroporosity • Plant available water capacity • Water infiltration	**Increase in:** • Acidification • Salinization • Sodication • Water pollution **Decrease in:** • Cation exchange capacity • Nutrient storage and cycling • Biogeochemical cycles	**Decrease in:** • Biomass production • Soil organic matter content • Nutrient content and cycling • Microbial biomass, activity, and diversity **Increase in:** • Organic matter decomposition • Eutrophication • Hypoxia • Emission of greenhouse gases

the source area. In the Sahara, a region in Africa with one of the highest wind erosion rates, dust emissions range between 400 and 700 Tg per year and are prone to increase with the projected change in climate (Washington et al., 2003).

A number of changes in physical, chemical, and biological processes occur due to the accelerated soil erosion (Table 1.2). These processes rarely occur individually but in interaction with one another (Eswaran et al., 2001). For example, compact soils are more prone to structural deterioration (physical process), salinization (chemical process), and reduced microbial activity (biological process) than un-compacted soils. Some processes are more dominant in one soil than in another. Salinization is often more severe in irrigated lands with poor internal drainage than in well-drained soils of favorable structure.

1.6 Drivers of Soil Erosion

Anthropogenic activities involving deforestation, overgrazing, intensive cultivation, soil mismanagement, cultivation of steep slopes, and urbanization accelerate the soil erosion hazard. Land use and management, topography, climate, and social, economic, and political conditions influence soil erosion (Table 1.3). In developing countries, soil erosion is directly linked to poverty level. Resource-poor farmers lack means to establish conservation practices. Subsistence agriculture forces farmers to use extractive practices on small size farm (0.5–2 ha) year after year for food production, delaying or completely excluding the adoption of conservation practices that reduce soil erosion risks (Lal, 2007). The leading three causes of accelerated soil erosion are: *deforestation, overgrazing, and mismanagement of cultivated soils.* About 35% of soil erosion is attributed to overgrazing, 30% to deforestation, and 28% to excessive cultivation (FAO, 1996).

Table 1.3 Factors affecting soil erosion and the attendant environmental pollution

Land Use	Cultivation	Climate and topography	Social and economic conditions
• Deforestation • Overgrazing • Urbanization • Slashing and burning • Mining • Industrial activities • Road constructions • Forest fires	• Excessive plow tillage • High chemical input • Irrigation • Salinization • Residue removal • Intensive row cropping • Monocropping • Shifting cultivation	• Frequent and intense droughts • Steep slopes (water and tillage erosion) • Rugged topography • Intense rainstorms • Frequent flooding • Intense windstorms • Flat terrains (wind erosion)	• Ineffective conservation policies • Poorly defined land tenure • Lack of incentives and weak institutional support • High population density • Low income • Non-availability of input

1.6.1 Deforestation

Forests provide essential ecosystem services such as soil erosion control, ecosystem stabilization, and moderation of climate and energy fluxes. Forests also provide wood, food, medicines, and many other wood-based products. Excessive logging and clear-cutting, expansion of agriculture to marginal lands, frequent fires, construction of roads and highways, and urbanization are the main causes of denudation. For example in Brazil alone, annually about 2.3 Mha of forest were removed between 1990 and 2000 (GEO, 2006). About 15 Mha yr^{-1} of forest are cleared annually worldwide and the rate of soil erosion is projected to accelerate with increase in deforestation (UNEP, 1997). Forests are disappearing more rapidly in developing than in developed countries (UN, 2005). Selective logging and shifting cultivation represent another 15 Mha of forest yr^{-1}. About half of the deforested areas are left bare or abandoned. Runoff and soil erosion rates are high from deforested areas. Deforestation removes the protective vegetal cover and accelerates soil erosion. In sloping lands, clearing of forest for agriculture can increase soil erosion by 5- to 20-fold (Benito et al., 2003).

1.6.2 Overgrazing

Herds of cattle and sheep are often concentrated on the same piece of land for too long in many livestock farms. This confinement results in overgrazing, repeated trampling or crushing, and soil displacement during traffic. Removing or thinning of grass reduces the protective cover and increases soil erosion particularly on steep slopes or hillsides. Overgrazing reduces soil organic matter content, degrades soil structure, and accelerates water and wind erosion. Trampling by cattle causes soil compaction, reduces root proliferation and growth, and decreases water infiltration rate and drainage. Increase in stocking rate results in a corresponding increase in runoff and soil erosion in heavily grazed areas. In wet and clayey soils, compaction and surface runoff from overgrazed lands can increase soil erosion. Increased erosion from pasturelands can also cause siltation and sediment-related pollution of downstream water bodies. In dry regions, animal traffic disintegrates aggregates in surface soils and increases soil's susceptibility to wind erosion. Continuous grazing increases the sand content of the surface soil as the detached fine particles are preferentially removed by flowing water and wind.

1.6.3 Mismanagement of Cultivated Lands

Expansion of agriculture to sloping, shallow, and marginal lands is a common cause of soil erosion. Intensive agriculture and plowing, wheel traffic, shifting cultivation, indiscriminate chemical input, irrigation with low quality water, and absence of vegetative cover degrade soils. Removal of crop residues for fodder and biofuel and industrial uses reduces the amount of protective cover left on the soil surface below the level adequate to protect the soil against erosion. Intensive cultivation accelerates water runoff and exacerbates soil erosion, which transport nutrients and pesticides off-site, declining soil and water quality. Shifting cultivation, a system in which depleted soils are abandoned to recover while new lands are cleared for cultivation, often worsens soil erosion as the duration of the fallow phase is reduced in densely populated regions. It often involves slashing and burning of forest or pasturelands to create new croplands, a common practice in tropical forests such as the Amazon. Cultivation is typically shifted after 3 yr, and the degraded soil is left in a short fallow cycle (2 or 3 yr), which does not provide long enough time for the soil to restore its functionality. Degraded soils require a longer period (5 to 40 yr) of time to fully recover. In some regions, because of the high population pressure and scarce arable land area, farmers are forced to use hilly, marginal or degraded lands for crop production.

1.7 Erosion in the USA

Among countries/ regions of the world, soil erosion is the lowest in the USA followed by that in Europe with a mean rate of 10 $\mathrm{Mg\,ha^{-1}\,yr^{-1}}$ (Pimentel, 2006). Indeed, models estimates show that water and wind erosion from croplands in the USA have decreased by about 35% between 1982 and 2003 (USDA-NRCS, 2007) (Fig. 1.5). The magnitude of decrease depends, however, on the region. Estimates show that rates of water erosion are the highest in Alabama (11.6 $\mathrm{Mgha^{-1}yr^{-1}}$) followed by Iowa, Georgia, and Mississippi, whereas those of wind erosion are the highest in New Mexico (28.9 $\mathrm{Mgha^{-1}yr^{-1}}$) followed by Colorado, Arkansas, and Texas (USDA-NRCS, 2007). Gains in erosion control can be significant in some areas but small or even negative in others because of the complexity of estimation, dynamic nature of soil erosion, and continuous changes in land use and management.

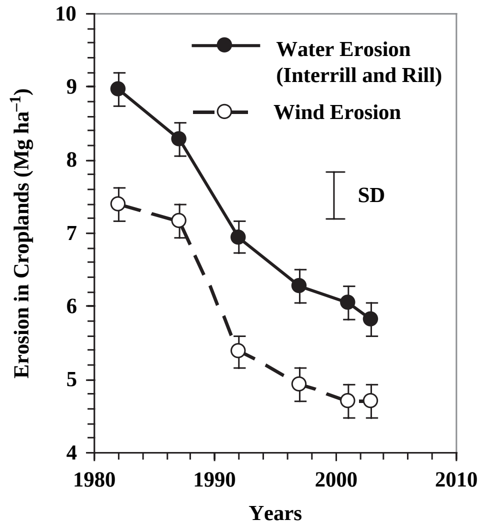

Fig. 1.5 Total soil erosion from croplands in the USA (After USDA-NRCS, 2007)

Results of modeling to estimate erosion risks may differ from those obtained by the point-specific measured data. Current estimates do not take into consideration the sediment and sediment-borne chemicals transported to downstream water bodies.

While the rates of total soil erosion have declined since 1970's, about one-third of U.S. croplands are eroding at rates faster than the tolerable rate, and that the rate of topsoil loss is 10 times faster than the rate of soil formation (Pimentel and Lal, 2007). Thus, the problem of soil erosion in the USA still persists. Erosion is particularly high in the major crop production areas under intensive tillage and monocropping. Soil-loss tolerance varies among soils and often ranges from 2.2 to 11.0 Mg ha^{-1} yr^{-1} (Troeh et al., 1999). Most of the prime agricultural lands are located in soils with an erosion tolerance level of 11.0 Mg ha^{-1} yr^{-1}. Some argue that the T values may be set too high and that even smaller rates of erosion can severely reduce crop production, depending on topsoil thickness and management systems. Soil erosion may gradually remove thin layers of soil of $\leq$1 mm thickness at a time. Even removal of 1 mm of soil, apparently very small, amounts to about 12.5 Mg ha^{-1}, which exceeds by far the rate of annual soil formation.

1.8 Global Distribution of Soil Erosion Risks

While soil erosion is not an imminent crisis in the USA and in other developed countries, the same can not be said about the impoverished regions of the world (Fig. 1.6). The problem of soil erosion is severe particularly in the tropics and sub-tropics because of the high population pressure, scarcity of prime agricultural lands, and predominance of resource-poor farmers. Soil erosion hazard has plagued mankind since the dawn of agriculture. Its magnitude and severity, however, increased during the 20th century due to population explosion and mismanagement of cultivated soils

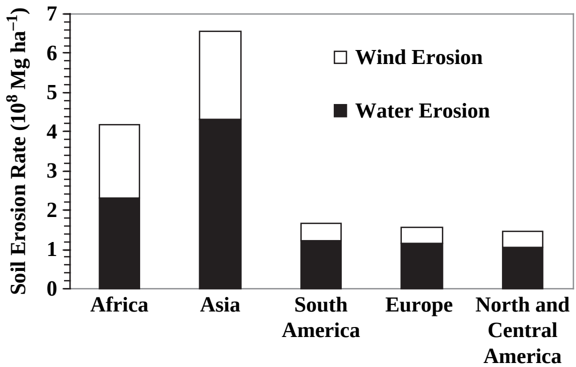

Fig. 1.6 Rates of soil erosion for selected continents (After WRI, 1992)

in Africa and South Asia (Kaiser, 2004) (Fig. 1.7). Erosion rates in these regions range from 30 and 40 Mg ha^{-1} yr^{-1} (Pimentel, 2006). Slash-and-burn agriculture for row cropping in marginal soils, sloping lands, and mountainous terrain is the main cause for the high rates of erosion.

Soil erosion contributes to the chronic malnutrition and rural poverty in the third world regions where farmers are too poor to establish erosion counteractive measures. The threat of erosion is region-specific. The main hot spots of soil erosion at present are: sub-Saharan Africa, Haiti, China Loess Plateau, the Andean region, the Caribbean (e.g., Haiti), and the lower Himalayas. The extent of soil degradation caused by deforestation, overgrazing, and poor soil management is the largest in Africa and Asia. On a global basis, soil erosion constitutes

Fig. 1.7 Map of Africa showing areas (*dark*) where soil degradation is a serious problem and population exceeds the land's carrying capacity (After Holden, 2006)

an ongoing problem. More attention is given to other agricultural topics than to soil erosion and its consequences. Pimentel (2000) lamented that soil erosion, while currently critical, is largely overlooked “because who gets excited about dirt?”.

At global scale, an estimate to about 1960 Mha of land are prone to erosion, which represents about 15% of the earth’s total land area, of which 50% is severely eroded, and much of that is being abandoned (Lal et al., 2004). Soil erosion rates ranges between 0.5 to 350 $Mg\,ha^{-1}\,yr^{-1}$. In some countries, about half of the agricultural prime lands are severely eroded. The current cultivated land area is almost equal to the land area abandoned since the dawn of agriculture. Annually, about 75×10^9 Mg of soil is lost worldwide, representing approximately US$400 billion per year for losses in nutrients, soil, and water, equivalent to US$70 per person per year (Lal, 1998). Soil erosion constitutes a major threat to food production particularly in densely populated and rapidly growing regions of the world. About $6 \times 10^9\,Mg\,yr^{-1}$ of soil is annually lost in India and China (Pimentel, 2006).

1.8.1 Soil Erosion in Africa and Haiti

The example of one of the most erosion-affected region in the world is Africa (Fig. 1.7). Soil erosion affects about one billion people globally, but about 50% of the affected population is concentrated in Africa. The total land area of Africa is about 30.2 million km^2 of which only 8.7 million km^2 (28.9%) is arable land (FAO, 2002a). Currently, 75% of the arable land in this continent is severely eroded (IFDC, 2006). Crop yields have been reduced by as much as 50% in the sub-Saharan Africa due to low nutrient input (Fig. 1.8), and excessive nutrient losses by erosion and crop extraction (Fig. 1.9). An average of 22 kg N (nitrogen), 3 kg P (phosphorus), and 15 kg K (potassium) ha^{-1} is lost annually (Eswaran et al., 2001). Crops are grown in the same piece of land year after year extracting large amounts of nutrients, which typically are not replenished by input of fertilizers and amendments due to the high cost and unavailability of fertilizers (Fig. 1.8). Since new lands for agricultural expansion are limited, as it was traditionally done (e.g., fallows), farmers are now forced to cultivate the same piece of land year after year and crop after crop. This continuous cropping with little or no nutrient input has induced overexploitation and severe mining of nutrients. Long fallows, while a norm in the past, have been replaced by short fallows or completely eliminated from agricultural systems due to land scarcity.

The continued downward spiral of nutrient depletion in Africa has resulted in sharp decline in crop yields. Average grain yield in most African countries is about 1 $Mg\,ha^{-1}$ which represents only 33% of the world average. The high rates of soil erosion and declining crop yields have increased problems of food insecurity and environmental degradation. Food production is either decreasing or remaining stagnant in most regions. Fertilizer use in the Sub-Saharan Africa is the lowest, corresponding to 10% of the world average (Fig. 1.8) (FAO, 2002b). Nutrients are

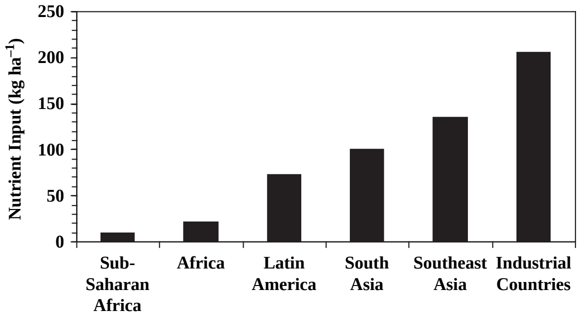

Fig. 1.8 Average annual use of nutrients in different parts of the world (After FAO, 2002b)

removed by harvested crops and animals (e.g., N), but in highly degraded soils most of the nutrients are removed by erosion and leaching. Low content of soil N is the main cause for the lower yields (Mafongoya et al., 2006). In some areas, deficiency in P and K is also evident. The high nutrient depletion is confounded by the low water retention capacity, compacted surface layers, low organic matter content, high acidity, and low aggregate stability of soils. Limited access to modern technologies such as inorganic fertilizers, improved crop varieties, and farm equipment has also contributed to nutrient mining. Deforestation confounds the problem as degraded soils are abandoned and new lands are cleared and intensively cultivated. About 50,000 ha of forest and 60,000 ha of grasslands are annually converted to extractive agriculture in Africa (IFDC, 2006).

Haiti, known as an eroding nation, is another example where soil erosion is very severe. Deforestation denudes mountains with disastrous consequences. About 97% of the previously forested lands have no trees, and about 30% of the deforested land is no longer arable (Kaiser, 2004). Most of the deforested lands are gullied with little or no topsoil left. Resource-poor farmers have no alternative but to cut trees for survival and farm steep slopes. The main adverse effect of erosion is on soil fertility and thus in reducing crop productivity in the region.

1.8.2 Drylands

Drylands or arid regions are most susceptible to degradation by wind erosion because of limited vegetative cover and harsh climate (e.g., low precipitation, strong winds) (Fig. 1.9). The total dryland area prone to degradation is about ∼3.6 billion ha, which represents about 60% of total dryland area in the world (UNEP, 1997). About 9–11 Mha of drylands are being abandoned annually (Daily, 1995). Rates of soil degradation in drylands are increasing steadily, particularly in developing nations. About 30% of the people in the world live in drylands where low productivity of crops and livestock is common.

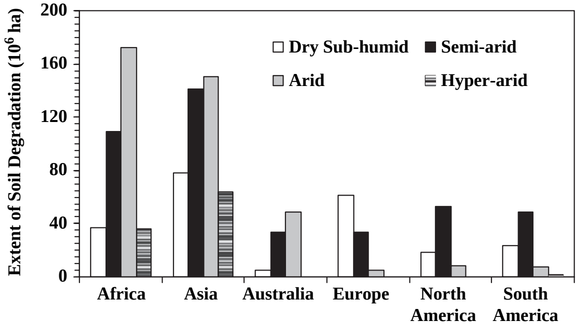

Fig. 1.9 Soil degradation in different ecological regions (After UNEP, 1992)

1.8.3 Magnitude of Wind Erosion

Erosion rates by wind in arid lands, which cover about 40% of the total land area in the world, can exceed those by water (Li et al., 2004). The Great Plains of the USA, Andes and The Pampas in South America, northern China, western Africa, and south-western Australia are regions where soil erosion by wind exceeds those by water. The “Dust Bowl” in the USA that occurred during the 1930’s is an illustration of the severity of wind erosion when proper soil conservation practices are not practiced.

Wind erosion has intensified in recent years due to the expansion of agriculture to marginal lands in developing countries. In China, for example, wind erosion affects about 20% of the total land area and is expanding rapidly due to intensive cultivation and grazing (Wang et al., 2006). Wind storms in northern China are eroding soil at 3600 $km^2\ yr^{-1}$ and in China Loess Plateau alone, soil erosion amounts to 1.6×10^9 Mg yr^{-1}. Wind erosion in the region is similar to that during the Dust Bowl era in the USA. Frequency of storm events in the region has increased since 1990’s and the resulting dust clouds are transported across oceans and continents. As an example, a severe dust storm that originated in a desert in western China on April 14, 1998 created immense clouds of dust which traveled over the Pacific and reached North America on April 27, 1998 (Shao, 2000). A large amount of wind storm dust is deposited in the oceans and a considerable portion reaches other continents.

Soil erosion by wind can be extremely high in arid and semiarid regions of the world. In the West African Sahel, one of the most severely affected regions by wind erosion in the world, annual wind erosion rates approach 200 Mg $ha^{-1}\ yr^{-1}$ from bare and highly erodible soils (Sterk, 2003). Intensively cultivated croplands in the region erode at a rate of 20–50 Mg $ha^{-1}\ yr^{-1}$, resulting in severe decline in crop yields (Bielders et al., 2000). Cultivation of poorly structured sandy and sandy loam soils with low organic matter content and fertility cause severe wind erosion in arid regions. In the semiarid region of Las Pampas in Argentina, rates of wind erosion range between 10 and 180 Mg $ha^{-1}\ yr^{-1}$ (Michelena and Irurtia, 1995). Soil erosion rates as high as 144 Mg $ha^{-1}\ yr^{-1}$ were reported from fallow fields in southern Alberta with erosion rates ranging from 0.3 to 30.4 Mg ha^{-1} per individual wind storm (Larney et al., 1995). In the USA, an average of about 25 cm of topsoil was lost by wind erosion between 1930 and 1950 in the Great Plains, representing approximately 156 $\mathrm{Mg\,ha^{-1}}$ of annual soil erosion (Chepil et al., 1952).

1.9 Current Trends in Soil and Water Conservation

Considerable progress has been made in developing conservation effective practices since the middle of the 20th century through a better understanding of causes, factors, and processes of soil erosion and the related soil properties. The understanding of the factors determining the magnitude of soil erosion risk has made possible the development and establishment of erosion control practices in many parts of the world. Despite these technological advances, the magnitude of soil erosion remains high.

The reasons for the decreasing trends in water and wind erosion rates in the USA since 1960's are linked to land stewardship and soil conservation efforts and policies. The Great Depression and the Dust Bowl that occurred during the 1930's have stirred interest and promoted research in developing soil conservation practices. Soil conservation policies were implemented in the early 1930's. The early policies stressed the importance of keeping the soil in place and were mostly focused on the on-site effects (e.g., crop production) of soil erosion. Since 1980's, conservation policies have stressed both on- and off-site adverse impacts of soil erosion. A number of USDA programs and initiatives exist that promote reduction in soil erosion and improvement in water quality and wildlife habitat. In 1985, the Food Security Act of 1985 created the Conservation Reserve Program (CRP) that compensates landowners and farmers for their land stewardship. The CRP provides technical and financial assistance to producers to implement approved conservation practices on highly erodible cropland. Adoption of no-till farming, a practice where crops are grown without turning soil, and conservation tillage have also contributed in part to the reduction of soil erosion. These efforts have resulted in better soil management, but much remains to be done. Water pollution with sediment and chemicals remains a major problem.

The significant improvements in soil and water conservation achieved in the USA and other developed countries are not reflected in the rest of the world where erosion constitutes a major threat to food security. More formidable measures of soil conservation are required to counteract soil erosion based on an integrated agronomic, economic, social, and political approach. Unless farming systems are based on economically feasible and environmentally sound practices of soil conservation, soil erosion poses a threat to agricultural and environmental sustainability. The magnitude and rate of soil erosion greatly vary with soil type, management, ecoregion, and climatic characteristics. Data on soil erosion in developing regions are extremely limited and estimates are crude particularly in erosion-prone and degraded areas. This is one of the reasons why some view that soil erosion crisis is exaggerated while others claim that soil erosion is serious and threatens the stability of agricultural production. Implications of erosion are either under- or over-estimated when credible data on the rate of erosion and its impact are non-existent or limited.

Summary

Water and wind erosion are the primary agents that cause soil erosion-induced degradation. Other causes of soil degradation include compaction, acidification, and salinization. Deforestation, overgrazing, intensive cultivation, mismanagement of cultivated soils, and urbanization are the main causes of accelerated soil erosion. Soil is eroding at rates faster than it is being formed and thus deserves more attention. Erosion is a global problem, but its magnitude is region-specific. Soil erosion has decreased in the USA since 1960's and ranges from 2.2 to 11 $\text{Mg ha}^{-1}\ \text{yr}^{-1}$, but that is still higher than the rate of soil formation. The problem of soil erosion in the rest of the world is more severe and erosion rates range between 30 and 40 $\text{Mg ha}^{-1}\ \text{yr}^{-1}$. The hot spots of soil erosion include the sub-Saharan Africa, Haiti, and the China Loess Plateau. About 15% of the earth's total land area is eroded, of which 50% is severely eroded and has been abandoned.

The on-site and offsite- impacts of accelerated soil erosion must be alleviated and managed to sustain agricultural productivity and environmental quality. Costs of erosion are high and affect the livelihood of all inhabitants particularly in poor regions of the world. Soil not only provides food security and maintains water resources clean but also affects the global climate. Soil is the medium that buffers water pollutants and stores C. Globally, soil erosion still remains a major issue. Technologies must be developed and proper conservation policies implemented in regions where soil erosion is the greatest risk and farmers are the poorest. Implementation of adequate conservation policies and programs have effectively stabilized or reduced soil erosion in developed countries but much more needs to be done. The needs are even greater in the developing regions of the world where economically deprived farmers do not have adequate resources to implement erosion control practices and mitigate the threat of soil erosion.

Study Questions

1. Describe the multi-functionality of the soil.
2. Describe the on-site and off-site impacts of soil erosion.
3. Briefly describe the history of soil erosion around the world.
4. What is the T value, and how is it estimated?
5. Discuss the uses and shortcomings of T value.
6. Soil erosion rates in the USA vary between 2.2 and 11.0 $\text{Mg ha}^{-1}\ \text{yr}^{-1}$. Convert these values to mm yr^{-1} assuming the soil bulk density of 1.25 Mg m^{-3}.
7. Three rainstorm events eroded 0.1, 0.3, and 0.5 mm of soil, respectively. Convert these values to Mg ha^{-1}, assuming the soil bulk density of 1.25 Mg m^{-3}.
8. Discuss results from Prob. 7 in relation to T values.
9. How can the erosion rates in Prob. 7 be reduced?
10. Discuss the soil processes affected by erosion.
11. Compare differences between water and wind erosion in terms of sediment transport.

12. Discuss the main reasons for the high rates of soil erosion in some regions and low in others.
13. What is the state-of-knowledge of soil erosion?

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Chapter 2
Water Erosion

Water erosion is the wearing away of the soil surface by water from rain, runoff, snowmelt, and irrigation. Rainwater in the form of runoff is the main driver of water erosion. It refers to the movement of soil organic and inorganic particles along the soil surface with flowing water and deposition of the eroded materials at lower landscape positions and in aquatic ecosystems. The eroded material can either form a new soil or simply fill lakes, reservoirs, and streams. Water erosion occurs in all soils to varying degrees. Slight erosion is actually beneficial to the formation of soil but severe or accelerated erosion adversely affects soil and environment. Understanding the mechanisms and magnitude of water erosion is vital to manage and develop erosion control practices. The goal of this Chapter is to describe the basic principles of water erosion including types, processes, factors, and causes.

2.1 Types

The main types of soil erosion are: *splash, interrill, rill, gully, streambank*, and *tunnel* erosion. Splash and sheet erosion are sometimes known as interrill erosion, but these two differ in the underlying fluvial processes.

2.1.1 Splash Erosion

Raindrops impacting the soil surface disperse and splash the soil, displacing particles from their original position. Splash erosion is caused by the bombardment of soil surface by impacting raindrops. Processes of splash erosion involve raindrop impact, splash of soil particles, and formation of craters (Ghadiri, 2004). Raindrops striking the soil surface develop a raindrop-soil particle momentum before releasing their energy in the form of splash. These raindrops strike the soil like small bombs forming craters or cavities of contrasting shapes and sizes. The depth of craters which is equal to the depth of raindrop energy penetration is a function of raindrop velocity, size, and shape.

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The formation of craters influences soil erosion. Mathematical relationships have been developed to estimate the crater characteristics as follows (Engel, 1961):

$$D = K R \left(\rho V^2\right)^{\frac{1}{3}} \tag{2.1}$$

where D is crater depth (cm), K is a constant, and R, ρ, and V are radius (cm), density (g cm^{-3}), and velocity (m s^{-1}) of raindrops, respectively. The crater volume (cm^3) and area (cm^2) are calculated as per Eq. (2.2) (Ghadiri, 2004):

$$V = \frac{1}{3}\pi D^2 \left(\frac{3d^2 + 12D^2}{8D} - D\right) \tag{2.2}$$

$$A = \pi \left(\frac{d^2}{4} + D^2\right) \tag{2.3}$$

where d is crater diameter (cm). The crater volume based on the raindrop kinetic energy (E) and bulk density (g cm^{-3}) of the soil (ρ_b) are estimated (Cook, 1959) using Eq. (2.4)

$$V = \frac{\rho_d{}^{\frac{1}{2}} \rho_b{}^{\frac{1}{2}}}{\rho_d + \rho_b} \times \frac{E_k}{2\phi} \tag{2.4}$$

where ρ_d is raindrop density, and ϕ is flow stress of the soil. The size of craters increases linearly with increase in raindrop energy. Understanding splash erosion is necessary to determine the process of soil erosion. Frequent splashing sculpts adjacent soil, rocks, stones, and vegetation over time.

2.1.2 Interrill Erosion

As soon as it starts, runoff promptly develops diminute rills, and that portion of runoff that flows between rills is called sheet or interrill erosion (Fig. 2.1). This type of erosion is mostly due to shallow flow. Some particles are carried away in runoff flowing in a thin sheet and some concentrate in small rills. Interrill is the most common type of soil erosion. Splash and interrill erosion make up about 70% of total soil erosion and occur simultaneously although splash erosion dominates during the initial process. Interrill erosion is a function of particle detachment, rainfall intensity, and field slope. It is represented as per Eq. (2.5) (Lane et al., 1987; Liebenow et al., 1990):

$$D_i = K_i \times I \times S \tag{2.5}$$

Fig. 2.1 A cropland affected by rill and interrill erosion (Courtesy USDA-NRCS)

where D_i is interrill detachment rate (kg m^{-2} s^{-1}), K_i is rate of interrill erodibility (kg m^{-2} s^{-1}), I is rainfall intensity (m s^{-1}), and S is slope factor which is equal to

$$S = 1.05 - 0.85 \exp(-4 \sin \theta) \tag{2.6}$$

where is θ slope angle.

2.1.3 Rill Erosion

It refers to the soil erosion that occurs in small channels or rills. Rill erosion occurs due to concentrated rather than shallow flow (Fig. 2.1). Runoff water that concentrates in small channels erodes soil at faster rates than interrill erosion. The force of flow and the soil particles creeping along the rill bed enlarge rills. Rill erosion is the second most common pathway of soil erosion. The rills are easily obliterated by tillage operations but can cause large soil erosion especially under intensive rains. Rill erosion is a function of soil erodibility, runoff transport capacity, and hydraulic shear of water flow. Soil erosion occurs mostly through the simultaneous action of interrill and rill erosion in accord with the steady-state sediment equation (Foster, 1982; Huang et al., 1996)

$$\frac{\partial q_s}{\partial_x} = D_r + Di \tag{2.7}$$

where q_s is sediment delivery rate in rills (kg m^{-1}s^{-1}), x is length of rill (m), D_r is rill detachment rate (kg m^{-2}s^{-1}), and D_i is interrill sediment delivery (kg m^{-2}s^{-1}). The D_r is computed as per Eq. (2.8):

$$D_r = \alpha\,(T_c - q_s) \tag{2.8}$$

where α is a constant and T_c is runoff transport capacity. After introducing a detachment capacity term, D_c, and leaving out D_i, the interrill erosion is represented as per Eq. (2.9)

$$D_c = \alpha T_c \tag{2.9}$$

$$\frac{\partial q_s}{\partial_x} = D_c\left(1 - \frac{q_s}{T_c}\right) \tag{2.10}$$

$$\frac{D_r}{D_c} + \frac{q_s}{T_c} = 1 \tag{2.11}$$

The D_r is equal to

$$D_c = K_r\,(\tau - \tau_c) \tag{2.12}$$

Replacing Eq. (2.12) into Eq. (2.10) results in

$$\frac{\partial q_s}{\partial_x} = K_r\,(\tau - \tau_c)\left(1 - \frac{q_s}{T_c}\right) \tag{2.13}$$

where K_r is rill erodibility (s m^{-1}), τ is hydraulic shear stress (Pa), and τ_c is critical shear stress (Pa). Eq. (2.13) reflects the intrinsic complex nature of erosion process.

2.1.4 Gully Erosion

Gully erosion creates either V- or U-shaped channels. The gullies are linear incision channels of at least 0.3 m width and 0.3 m depth. Gullies are primarily formed by concentrated runoff converging in lower points of the field (Fig. 2.2). Thus, erosion occurring in these channels is known as concentrated flow erosion. Undulating fields cause runoff to concentrate in natural swales as runoff moves downslope in narrow paths in the form of channelized flow. Continued gully erosion removes entire soil profiles in localized segments of the field. As gullies grow, more sediment is transported.

2.1.4.1 Types

There are two types of gullies: *ephemeral* and *permanent*. *Ephemeral gullies* are shallow channels that can be readily corrected by routine tillage operations. In contrast, *permanent gullies* are too large to be smoothed by regular tillage or crossed

Fig. 2.2 Concentrated runoff forms gullies (Courtesy USDA-NRCS). Channels without hydraulic roughness elements erode at faster rates with incoming runoff than those nested with deep plant roots and rocks. Gullies are expanded by steep water fall at the gully heads, called headcut, and by gradual lateral erosion and sloughing of the gully sides

by machinery traffic and require expensive measures of reclamation and control. Ephemeral gullies following removal tend to reform in the same points of the field if not controlled. Even if gullies are repaired by tillage, soil is already lost as the eroded material is transported off-site. Gullies are normally back filled with soil from neighboring fields which reduces the topsoil depth.

2.1.4.2 Factors

The shear stress of flowing water and critical shear stress of the soil are two prominent factors affecting gully erosion. The shear stress of flow is responsible for continued detachment of channel bed and sides and transport of eroded materials along the well defined ephemeral channels. Equation (2.13) also applies to gully erosion when rills are scaled up to larger channels. Grassed waterways reduce gully formation, but when the flow shear stress exceeds the critical stress of the soil and plant roots, the cover fails and shear stress of the flow rapidly increases, enlarging the gullies and causing severe soil erosion. Bare and freshly plowed soils have the lowest critical shear stress and thus are the most susceptible to gully erosion. Critical shear of soil is a function of soil texture, bulk density, clay content, dispersion ratio, tillage, plant roots, residue cover, and soil slope.

$$\text{Shear stress of runoff} < \text{Critical shear of soil} = \text{No gully formation}$$
$$\text{Shear stress of runoff} > \text{Critical shear of soil} = \text{Gully formation}$$

The widening of an ephemeral gully with successive rain storms can be expressed as per Eq. (2.14) (Foster and Lane, 1983):

$$\Delta W = \left[1 - \exp\left(-t_*\right)\right]\left(W_f - W_i\right) \tag{2.14}$$

where ΔW is change in channel width, W_f is final channel width under the new storm, W_i is initial channel width, and t_* is time.

$$t_* = \frac{t\left(\frac{\partial W}{\partial t}\right)_i}{\left(W_f - W_i\right)} \tag{2.15}$$

where $\left(\frac{\partial W}{\partial t}\right)_i$ is initial rate of change in channel width with respect to the previous width. A rapid approximation of the amount of soil eroded by gully erosion is done by measuring the size of the gully (length and area) and correlating it with the bulk density of the reference soil (Foster, 1986). This simple approach can be related to the whole landscape by the voided area with reference to the uneroded portions of the fields. Advanced techniques of mapping gully erosion across large areas involve aerial photographs, remote sensing, and geographic information systems (GIS) tools. Conservation practices such as no-till, reduced tillage, and residue mulch are effective to control rill and interrill erosion but not gully erosion. Permanent grass waterways, terraces, and mechanical structures (e.g., concrete structures) are often used to control gully erosion (See Chapter 11).

2.1.5 Tunnel Erosion

Tunnel erosion, also known as pipe erosion, is the underground soil erosion and is common in arid and semiarid lands. Soils with highly erodible and sodic B horizons but stable A horizons are prone to tunnel erosion. Runoff in channels, natural cracks, and animal burrows initiates tunnels by infiltrating into and moving thorough dispersible subsoil layers. The surface of tunnel erosion-affected soils is often stabilized by roots (e.g., grass) intermixed with soil while the subsoil is relatively loose and easily erodible. Presence of water seepage, lateral flow, and interflow is a sign of tunnel erosion. The tunnels or cavities expand to the point where they no longer support the surface weight and collapse forming potholes and gullies. Tunnel erosion changes the geomorphic and hydrologic characteristics of the affected areas. Reclamation procedures include deep ripping, contouring, revegetation with proper fertilization and liming, repacking and consolidation of soil surface, diversion of concentrated runoff, and reduction of runoff ponding. Revegetation must include trees and deep rooted grass species to increase water absorption.

2.1.6 Streambank Erosion

It refers to the collapse of banks along streams, creeks, and rivers due to the erosive power of runoff from uplands fields (Fig. 2.3). Pedestals with fresh vertical cuts along streams are the result of streambank erosion. Intensive cultivation, grazing,

Fig. 2.3 Corn field severely affected by streambank erosion (Courtesy USDA-NRCS). Saturated soils along streambanks slump readily under concentrated runoff, which causes scouring and undercutting of streambanks and expansion of water courses

and traffic along streams, and absence of riparian buffers and grass filter strips accelerate streambank erosion. Planting grasses (e.g., native and tall grass species) and trees, establishing engineering structures (e.g., tiles, gabions), mulching stream borders with rocks and woody materials, geotextile fencing, and intercepting/diverting runoff are measures to control streambank erosion.

2.2 Processes

Water erosion is a complex three-step natural phenomenon which involves *detachment*, *transport*, and *deposition* of soil particles. The process of water erosion begins with discrete raindrops impacting the soil surface and detaching soil particles followed by transport. Detachment of soil releases fine soil particles which form surface seals. These seals plug the open-ended and water-conducting soil pores, reduce water infiltration, and cause runoff. At the microscale level, a single raindrop initiates the whole process of erosion by weakening and dislodging an aggregate which eventually leads to large-scale soil erosion under intense rainstorms. The three processes of erosion act in sequence (Table 2.1).

The first two processes involving dispersion and removal of soil define the amount of soil that is eroded, and the last process (deposition) determines the distribution of the eroded material along the landscape. If there were no erosion, there would be no deposition. Thus, detachment and entrainment of soil particles are the primary processes of soil erosion, and, like deposition, occur at any point of soil.

Table 2.1 Role of the three main processes of water erosion

Detachment	Transport	Deposition
• Soil detachment occurs after the soil adsorbs raindrops and pores are filled with water. • Raindrops loosen up and break down aggregates. • Weak aggregates are broken apart first. • Detached fine particles move easily with surface runoff. • When dry, detached soil particles form crusts of low permeability. • Detachment rate decreases with increase in surface vegetative cover.	• Detached soil particles are transported in runoff. • Smaller particles (e.g., clay) are more readily removed than larger (e.g., sand) particles. • The systematic removal of fine particles leaves coarser particles behind. • The selective removal modifies the textural and structural properties of the original soil. • Eroded soils often have coarse-textured surface with exposed subsoil horizons. • Amount of soil transported depends on the soil roughness. • Presence of surface residues and growing vegetation slows runoff.	• Transported particles deposit in low landscape positions. • Most of the eroded soil material is deposited at the downslope end of the fields. • Placing the deposited material back to its origin can be costly. • Runoff sediment transported off-site can reach downstream water bodies and cause pollution. • Runoff sediment is deposited in deltas along streams. • Texture of eroded material is different from the original material because of the selective transport process.

When erosion starts from the point of raindrop impact, some of the particles in runoff are deposited at short distances while others are carried over long distances often reaching large bodies of flowing water.

2.3 Factors

The major factors controlling water erosion are *precipitation, vegetative cover, topography, and soil properties* and are discussed in Table 2.2. The interactive effects of these factors determine the magnitude and rate of soil erosion. For example, the longer and steeper the slope, the more erodible the soil, and the greater the transport capacity of runoff under an intense rain. The role of vegetation on preventing soil erosion is well recognized. Surface vegetative cover improves soil's resistance to erosion by stabilizing soil structure, increasing soil organic matter, and promoting activity of soil macro- and micro-organisms. The effectiveness of vegetative cover depends on plant species, density, age, and root and foliage patterns.

2.4 Agents

Two main agents affecting soil erosion by water are: rainfall and runoff erosivity.

Table 2.2 Factors affecting water erosion

Climate	Vegetative cover	Topography	Soil properties
• All climatic factors (e.g., precipitation, humidity, temperature, evapotranspiration, solar radiation, and wind velocity) affect water erosion. • Precipitation is the main agent of water erosion. • Amount, intensity, and frequency of precipitation determine the magnitude of erosion. • Intensity of rain is the most critical factor. • The more intense the rainstorm, the greater the runoff and soil loss. • High temperature may reduce water erosion by increasing evapotranspiration and reducing the soil water content. • High air humidity is associated with higher soil water content. • Higher winds increase soil water depletion and reduce water erosion.	• Vegetative cover reduces erosion by intercepting, adsorbing, and reducing the erosive energy of raindrops. • Plant morphology such as height of plant and canopy structure influences the effectiveness of vegetation cover. • Surface residue cover sponges up the falling raindrops and reduces the bouncing of drops. It increases soil roughness, slows runoff velocity, and filters soil particles in runoff. • Soil detachment increases with decrease in vegetative cover. • Dense and short growing (e.g., grass) vegetation is more effective in reducing erosion than sparse and tall vegetation. • The denser the canopy and thicker the litter cover, the greater is the splash erosion control, and the lower is the total soil erosion.	• Soil erosion increases with increase in field slope. • Soil topography determines the velocity at which water runs off the field. • The runoff transport capacity increases with increase in slope steepness. • Soils on convex fields are more readily eroded than in concave areas due to interaction with surface creeping of soil by gravity. • Degree, length, and size of slope determine the rate of surface runoff. • Rill, gully, and stream channel erosion are typical of sloping watersheds. • Steeper terrain slopes are prone to mudflow erosion and landslides.	• Texture, organic matter content, macroporosity, and water infiltration influence soil erosion. • Antecedent water content is also an important factor as it defines the soil pore space available for rainwater absorption. • Soil aggregation affects the rate of detachment and transportability. • Clay particles are transported more easily than sand particles, but clay particles form stronger and more stable aggregates. • Organic materials stabilize soil structure and coagulate soil colloids. • Compaction reduces soil macroporosity and water infiltration and increases runoff rates. • Large and unstable aggregates are more detachable. • Interactive processes among soil properties define soil erodibility.

2.5 Rainfall Erosivity

It refers to the intrinsic capacity of rainfall to cause soil erosion. Water erosion would not occur if all rains were non-erosive. Since this is hardly the case, knowledge of rainfall erosivity is essential to understanding erosional processes, estimating soil erosion rates, and designing erosion control practices. Properties affecting erosivity are: *amount*, *intensity*, *terminal velocity*, *drop size*, and *drop size distribution* of rain (Table 2.3). These parameters affect the total erosivity of a rain, but measured data are not always available in all regions for an accurate estimation of rain erosivity. Erosivity of rain and its effects differ among climatic regions. The same amount of rain has strikingly different effects on the amount of erosion depending on the intensity and soil surface conditions. Rains in the tropics are more

Table 2.3 Factors affecting the erosivity of rainfall

Amount	Intensity	Terminal velocity	Drop size
• More rain results in more erosion although this correlation depends on rainfall intensity. • Amount of rain is a function of duration and intensity of rain. • Measurement of the amount of rain is influenced by the type, distribution, and installation protocol of the rain gauges. • Height of rain gauges and wind drift affect measurement. • Available measured data are only point estimates of a large area.	• Intensity is the amount of rain per unit of time (mm h^{-1}). • Intensity is normally <70 mm h^{-1} in temperate regions, but it can be as high as 150 mm h^{-1} in tropical regions. • Intense storms are often of short duration. • Intensity is directly correlated with erosion. • The more intense the rain, the greater is the soil erosion. • Many erosion models use kinetic energy based on rain intensity. • Intensity is obtained from daily rain gauges with charts and computerized systems.	• A raindrop accelerates its velocity until the air resistance equals the gravitational force, and then it falls at that constant velocity, also known as terminal velocity. • Raindrops can strike the soil at a speed as high as 35 km h^{-1} and displace soil particles as far as 2 m in horizontal and 1 m in vertical direction. • Terminal velocity increases with increase in raindrop size. • Faster falling large raindrops have more erosive power than smaller drops. • Raindrops of 5 mm in diameter have a terminal velocity of about 9 m s^{-1}	• Size of raindrops can range between 0.25 and 8 mm in diameter, but those between 2 and 5 mm are common. • In intense storms, raindrops can be as large as 8 mm. • While drop size increase with increase in rain intensity, it may decrease when intensities exceed 100 mm h^{-1}. • Drop-stain (use of absorbent paper with water-soluble dyes) and flour-pellet (collecting and drying drops in a container with flour) are methods used for measuring raindrop size distribution, along with radar and imaging techniques.

erosive than those in temperate regions due to the presence of strong winds and high temperature. Annual distribution of rainfall also influences the erosivitiy of rain. Rains in temperate regions are uniformly distributed across seasons, known as *unimodal*, and cause less erosion than those intense rains in tropical regions, which are distributed in two seasons, known as *bimodal*.

Intensity is the most important rainfall property that determines the amount of erosion (Table 2.3). Combination of high amount with high intensity of rain produces high erosion. Intense storms are of short duration but cause large amounts of erosion. The total intensity of a storm is made up of the intensity of individual raindrops. The energy of a raindrop due to its motion, known as kinetic energy, is a function of the raindrop size and its terminal velocity. The kinetic energy (E) in ergs of a falling raindrop is estimated as:

$$E = \frac{1}{2} m V^2 \tag{2.16}$$

where m is the mass of falling raindrop (g), and v is the velocity of fall (cm s^{-1}). The total kinetic energy for the storm can be estimated by summation of E values from individual raindrops. Measurement of E of raindrops is difficult under natural rain. Electronic sensors based on optical and laser devices have been used for direct measurements (Lovell et al., 2002). When a raindrop impacts sensors, it produces sound waves which are converted to measurable scales. Simple raindrop techniques are used to study the E of raindrops impacting individual soil aggregates and causing soil erosion.

Several mathematical relationships exist to relate intensity to the total energy of rainfall. The Universal Soil Loss Equation (USLE) and the revised USLE use, for example, data on rain intensity to compute E and then compute the total kinetic energy of the storm (Wischmeier and Smith, 1978) as follows:

$$E = 0.119 + 0.0873 Log_{10}(i_m) \qquad i_m \leq 76\,\text{mm h}^{-1} \tag{2.17}$$

$$E = 0.283 \qquad i_m > 76\,\text{mm h}^{-1} \tag{2.18}$$

where E is in megajoule ha^{-1} mm^{-1} of rainfall, and i_m is rainfall intensity (mm h^{-1}). When rainfall is measured in daily totals, the E in USLE is estimated as a function of the rainfall depth (D) (mm) and intensity (i) (mm h^{-1}) of rainfall as follows:

$$E = \frac{D\,(210 + 89 Log_{10} i)}{100} \tag{2.19}$$

The i for rainfall events of different return periods required for designing erosion control practices can be represented as

$$i = \frac{K T^x}{t^n} \tag{2.20}$$

where K, x, and n are constants specific to a location, t is the storm duration (min), and T is the return period (yr). Rainfall frequency data including rain duration from 30 min to 24 h and return periods from 1 to 100 yr are available for the USA (Hershfield, 1961).

2.6 Runoff Erosivity

Runoff, also known as overland flow or surface flow, is the portion of water from rain, snowmelt, and irrigation that runs off the field and often reaches downstream water courses or bodies such as streams, rivers, and lakes. Runoff occurs only after applied water: (1) is absorbed by the soil, (2) fills up the soil pores and surface soil depressions, (3) is stored in surface detention ponds if in place, and (4) accumulates on the soil surface at a given depth. The components of water balance for runoff to occur are:

$$\begin{aligned}\text{Runoff} &= \text{INPUT} - \text{OUTPUT}\\ &= (Rain, Snowmelt, Irrigation) - (Infiltration, Evaporation, Rain\\ &\quad\; Interception\ by\ Canopy, Water\ \ Absorption, Transpiration,\\ &\quad\; Surface\ Detention)\end{aligned}$$

Similar to the rainfall erosivity, runoff erosivity is the ability of runoff to cause soil erosion. Raindrops impacting soil surface loosen up, detach, and splash soil particles, while runoff carries and detaches soil particles. Interaction among rain, runoff, and soil particles results in erosion. Floating and creeping soil particles in turbulent runoff also contribute to aggregate detachment. Rain has more erosive power than runoff.

The kinetic energy (E) of a rain of mass equal to m and terminal velocity (v) equal to 8 m s^{-1} is (Hudson, 1995)

$$E = \frac{1}{2}m(8)^2 = 32m \tag{2.21}$$

Assuming that 25% of the rain becomes runoff and the runoff velocity is 1 m s^{-1}, the E of runoff is

$$E = \frac{1}{2}\left(\frac{m}{4}\right)(1)^2 = \left(\frac{1}{8}\right)m \tag{2.22}$$

Thus, the E of rain is 256 times greater than that of runoff.

If 50% of the rain had become runoff, the E would be greater by 128 times. Even if all the rain had become runoff, the rain would still have greater E because of the greater terminal velocity of the rain.

The capacity of runoff to scour the soil and transport particles increases with runoff amount, velocity, and turbulence. Runoff carries abrasive soil materials which further increase its scouring capacity. Early erosion models such as the USLE considered only rainfall erosivity. Improved models which partition the erosive force of water in rainfall erosivity and runoff erosivity provide more accurate predictions. One such relationship which accounts for both components is the modified USLE (MUSLE) (Foster et al., 1982) represented as:

$$R_e = 0.5EI_{30} + \alpha 0.5 Q_e {q_p}^{0.33} \tag{2.23}$$

where R_e is the rainfall-runoff erosivity, EI_{30} is product of rain E and its 30-min. intensity (I_{30}) of the USLE (MJ. mm ha^{-1} h^{-1}), is α is a coefficient, Q_e is the runoff depth (mm), and q_p is the peak runoff rate (mm h^{-1}).

2.6.1 Estimation of Runoff

The determination of the maximum runoff rate and total amount of runoff leaving a watershed are of great utility to:

- design and construct mechanical structures of erosion control (e.g., ponds, terraces, channels),
- design and establish conservation buffers (e.g., grass barriers, vegetative filter strips, riparian buffers),
- estimate the probable amount of sediment and chemicals (e.g., fertilizers, pesticides) transport in runoff, and
- convey runoff water safely in channels or grass waterways at a reduced erosive power.

Determining rate and volume of runoff involves the consideration of the various runoff factors such as topography, soil surface conditions (e.g., roughness), soil texture, water infiltration, and vegetative cover. When rain falls on an impermeable surface such as a paved surface, all the rain becomes runoff. This is not the case under natural soil conditions where rainfall is partitioned into various pathways: interception by plants and surface residues, infiltration, evaporation, accumulation in surface depressions, and runoff. Any mathematical equation that attempts to estimate runoff from a watershed must consider all these factors.

2.6.2 Time of Concentration

Time of concentration is the time required for the runoff water to travel from the farthest point in terms of travel time to the outlet of the watershed (Schwab et al., 1993). Assume that a rain falls only at the lower end of a watershed. Such being the case, runoff water from a point near the upper end of the wetted portion would reach the outlet of the watershed in a shorter time than that from the most distant point of the watershed if it rained in the whole watershed. The greatest amount of runoff results when the whole watershed is contributing to runoff under the same rainfall intensity. The time that it takes for the whole watershed to produce runoff depends on the time of concentration. The longest time may not always correspond to the most distant point from the outlet as variability in surface roughness (e.g., major depressions) even near the outlet could delay the time for the water flow to reach the outlet.

The time of concentration is critical to compute the runoff hydrograph. The shape and peak of runoff rate are a function of runoff travel time in all its forms including interrill and rill flow. Development of impervious surfaces in urban areas dramatically decreases the time of concentration and increases the peak discharge rates. The time of concentration primarily depends on the following factors:

2.6.2.1 Surface Roughness

The smoother the surface of a watershed, the smaller is the time of concentration. Growing vegetation, residue mulch, rock outcrops, ridges, depressions, and other obstacles retard the overland flow. Thus, travel time in a vegetated watershed is increased unless the flow is conveyed in constructed channels, which conduct water more rapidly. Surface roughness is expressed in terms of Manning's roughness coefficient, which varies according to the type of obstacles (Table 2.4).

Table 2.4 Manning's coefficient of roughness for selected surface conditions (After Engman, 1986)

Condition of the soil surface	Manning's coefficient (n)
Bare soil	0.011
Impervious surface (paved surfaces)	0.011
Continuous fallow without residue	0.05
Cultivated soil with ≤20% of residues	0.06
Cultivated soil with ≥20% of residues	0.17
Short grass prairie	0.15
Tall and dense grass prairie including native species (weeping lovegrass, bluegrass, buffalo grass, switchgrass, Indian grass, and big bluestem).	0.24
Trees	0.40–0.80

2.6.2.2 Watershed Slope

The steeper the surface of a watershed, the shorter the time that it takes for water to reach the outlet. Terracing and establishment of conservation buffers reduce the watershed slope and thereby increase the travel time of water flow. In urban areas, grading changes the slope. Channels with reduced roughness increase runoff velocity and peak discharge. On the contrary, establishment of ponds and reduction of soil slope increase the time of concentration.

2.6.2.3 Size of the Watershed

The larger the watershed, the greater the contributing area to runoff but longer the time for runoff to travel (Fig. 2.4). Both size and shape of the watershed influence the travel time of runoff. Runoff rate reaches its peak faster in a shorter than a longer watershed.

2.6.2.4 Length and Shape the Channel

Water flow from the farthest point in flow time under field conditions is not always laminar but tends to flow in different ways including through: (1) shallow rills, (2) open channels as concentrated flow, and (3) diffuse interrill flow. After a short distance, interrill or sheet flow becomes concentrated flow in channels. The longer and smoother the channel, the shorter is the travel time to reach the outlet. Sloping and straight channels accelerate the runoff velocity. Channels that are straightened out increase runoff velocity as compared to meandering and tortuous channels.

The common equation to compute the time of concentration is that developed by Kirpich (1940):

$$T_c = 0.0195 L^{0.77} S^{-0.385} \tag{2.24}$$

where T_c is time of concentration (min), L is maximum length of flow (m), and S is slope of the watershed (m m^{-1}). Rainfall duration can be higher, lower, or equal to the time of concentration.

The time of concentration for overland and channel flow is computed by summing up both types of flow time as (USDA-SCS, 1986):

$$T_c = t_{\text{ov}} + t_{ch} \tag{2.25}$$

where t_{ov} is time of concentration for overland flow (min) and t_{ch} is time of concentration for channel flow (min).

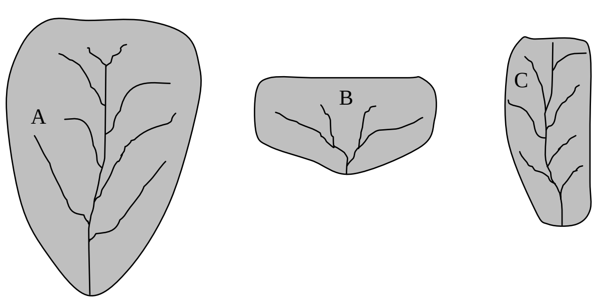

Fig. 2.4 A large watershed under both overland and channel flow (**A**) and two watersheds (**B** and **C**) of the same size but oriented differently, yielding thus different times of concentration (After Hudson, 1995). Size, shape, and orientation of the watershed influence the runoff travel time and peak runoff rates

$$t_{\mathrm{ov}} = \frac{L^{0.6} \times n^{0.6}}{18S^{0.3}} \tag{2.26}$$

$$t_{ch} = \frac{0.62 \times L_{ch} \times n_{ch}^{0.75}}{A^{0.125} \times S_{ch}^{0.375}} \tag{2.27}$$

where L is slope length of the watershed (m), n is Manning's roughness coefficient for the watershed, S is average slope gradient of the watershed (m m^{-1}), L_{ch} is channel length from the farthest point in flow time (km), n_{ch} is Manning's roughness coefficient for the channel, A is area of the watershed (km^2), and S_{ch} is slope of the channel (m m^{-1}). In topographically complex watersheds with a large network of channels, the concentration is estimated for each segment in the watershed as:

$$T_c = T_{c1} + T_{c2} + T_{c3} + \ldots\ldots\ldots\ldots T_{cn} \tag{2.28}$$

where T_{c1}, T_{c2}, T_{c3}, and T_{cn} are time of concentration for watershed segments 1, 2, and 3, respectively, and n is the number of flow segments.

Example 1. Estimate the time of concentration for a watershed of 1.5 km^2 that has an overland slope length of 80 m with a slope of 5.5%. The channel length is 6 km with a slope of 0.9%. The Manning's coefficient of roughness for the watershed is 0.15 and that for the channel is 0.014.

$$t_{\mathrm{ov}} = \frac{L^{0.6} \times n^{0.6}}{18S^{0.3}} = \frac{(80)^{0.6} \times (0.15)^{0.6}}{18 \times (0.055)^{0.3}} = \frac{4.44}{7.54} = 0.589\,h$$

$$t_{ch} = \frac{0.62 \times L_{ch} \times n_{ch}^{0.75}}{A^{0.125} \times S_{ch}^{0.375}} = \frac{0.62 \times 6\,km \times (0.014)^{0.75}}{(1.5\,km)^{0.125} \times (0.009)^{0.375}} = \frac{0.151}{0.180} = 0.839\,h$$

The time of concentration for both types of flow is:

$$T_c = t_{ov} + t_{ch} = 0.589 + 0.839 = 1.428\,\mathrm{h}.$$

2.6.3 Runoff Volume

The total amount of runoff leaving a field can be computed using the runoff curve number (CN) method, an empirical approach widely used to compute runoff volume for different soil types and surface conditions, as follows:

$$Q = \frac{\left(R_{day} - I_a\right)^2}{\left(R_{day} - I_a\right) + S} \tag{2.29}$$

where Q is depth of runoff (mm), R_{day} is amount of rainfall (mm) for the day, I_a is initial abstraction that accounts for the surface water storage in depressions or ponding, rainfall interception by plants and litter/residues, evaporation, and infiltration before runoff starts (mm), and S is retention parameter (mm). The I_a, a complex parameter, depends on soil surface and vegetative cover characteristics and is assumed to be equal to:

$$I_a = 0.2S \tag{2.30}$$

Substituting Eq. (2.30) in Eq. (2.29) results in

$$Q = \frac{\left(R_{day} - 0.2S\right)^2}{R_{day} + 0.8S} \tag{2.31}$$

Thus, S becomes the parameter which accounts for the differences in soil surface conditions, land use and management, and antecedent water content. It reflects the land use conditions through the CN, which is equal to:

$$S = \frac{25400}{CN} - 254 \tag{2.32}$$

Among the factors that influence CN are hydrologic soil group, land use, soil management, cropping system, conservation practices, and antecedent water content. The values of CN vary from 0 to 100 depending on the soil and surface conditions (Table 2.5). Values of CN decrease with increase in surface vegetative cover. Bare soils without crop residues have the largest CN values whereas undisturbed soils covered by dense vegetation have the smallest CN values. Soils based on their infiltration characteristics and runoff potential are classified into four main hydrologic groups: A, B, C, and D. A hydrologic soil group refers to a group of soils having the same runoff potential under similar rainstorms and surface cover conditions. Important factors which determine the runoff potential include infiltration capacity, drainage, saturated hydraulic conductivity, depth to water table, and presence of impermeable layer.

2.6.4 Characteristics of the Hydrologic Groups

A: These soils are deep, highly permeable, and their textural class includes sand, loamy sand, and sandy loam. Because of the low clay content, soils in this group have very high saturated hydraulic conductivity and infiltration rates even when completely wet and thus have the *lowest runoff potential*. Deep loess and sandy soils are part of this group.

B: This group includes silt loam and loamy soils, which are moderately deep and permeable. They transmit water at slightly lower rates than group A although the

Table 2.5 Runoff curve numbers for selected surface conditions for different soil hydrologic groups (After USDA-SCS, 1986)

Surface condition	Hydrologic condition	Hydrologic soil group			
		A	B	D	C
Urban Areas					
Impervious areas (roofs, streets, parking lots, and driveways)		98	98	98	98
Pervious areas (lawns, parks, golf courses, etc.)	Good	39	61	74	80
Gravel streets and roads		76	85	89	91
Compacted soil surface (roads and streets and right-of-way)		72	82	87	89
Agricultural Lands					
Fallow: Bare soil		77	86	91	94
Fallow: Crop residue cover	Poor	76	85	90	93
	Good	74	83	88	90
Row crops 1. Straight rows	Poor	72	81	88	91
	Good	67	78	85	89
2. Straight rows + residue cover	Poor	71	80	87	90
	Good	64	75	82	85
3. Straight rows + contoured and	Poor	65.	73	7980	81
terraced + residue cover	Good	61	70	77	80
Small grains:					
Straight rows	Poor	65	76	84	88
	Good	63	75	83	87
Straight rows + residue cover	Poor	64	75	83	86
	Good	63	75	83	87
Straight rows + contoured and	Poor	60	71	78	81
terraced + residue cover	Good	58	69	77	80
Legumes or crop rotations + contoured and	Poor	63	73	80	83
terraced	Good	51	67	76	80
Non-Cultivated Lands					
Pasturelands, grasslands, and	$<$50% ground cover	68	79	86	89
rangelands	50% to 75% cover	49	69	79	84
	$>$75% cover	39	61	74	80
Woods	Grazed or regularly burned	45	66	77	83
	Grazed but not burned	36	60	73	79
	Ungrazed	30	55	70	77

rates are still above the average values. The moderate permeability results in soils with *moderately low runoff potential*.

C: These soils are less permeable and shallower than those in group B because of relatively high clay content or presence of slowly permeable layers below the topsoil. Sandy clay loams are within this group. These soils have moderately high runoff potential due to the low rates of water transmission.

D: This group comprises clay loam, silty clay loam, sandy clay, silty clay or clay. It includes soils with nearly impermeable layers (e.g., claypan) and with shallow water table. These soils have very low infiltration rates and saturated hydraulic conductivity and have the highest runoff potential.

Example 2. Estimate the runoff amount that is produced by a watershed of 2 km^2 receiving an average precipitation of 50 mm per day. The watershed is under three different uses. Half of the watershed consists of agricultural lands with crops planted in straight rows under good condition, a third of the watershed consists of residential area with 30% of impervious surface from houses and paved driveways, and the rest of the watershed is under woods with dense litter cover. The soils are part of the hydrologic group B.

Solution.

Impervious area: $0.30 \times 98 = 29.4$
Pervious area: $0.70 \times 61 = 42.7$

Land use	Fraction of area	Average curve number	Weighted curve number
Row crops in good condition	0.500	78	39.0
Residential Area	0.333	$29.4 + 42.7 = 72.1$	24.0
Woods	0.167	55	9.2
			Total = 72.2

Compute S using the weighted CN value:

$$S = \frac{25400}{CN} - 254 = \frac{25400}{72.18} - 254 = 97.9\,\text{mm}$$

Next, compute runoff depth:

$$Q = \frac{\left(R_{day} - 0.2S\right)^2}{R_{day} + 0.8S} = \frac{(50\,\text{mm} - 0.2 \times 97.9)^2}{(50\,\text{mm} + 0.8 \times 97.9)} = \frac{925.4\,\text{mm}^2}{128.3\,\text{mm}} = 7.2\,\text{mm}$$

Runoff in terms of volume is computed as:

$$Q = 2\,\text{km}^2 \times \frac{(1000\,\text{m})^2}{1\,\text{km}^2} \times 0.00717\,\text{m} = 14{,}340\,\text{m}^3$$

2.6.5 Peak Runoff Rate

The peak runoff rate is the maximum rate of runoff that occurs during a rainfall event.

It is estimated using the rational method and the modified rational method. The rational method is as follows:

$$q = \frac{C \times i \times A}{3.6} \tag{2.33}$$

where q is peak runoff rate ($m^3\ s^{-1}$), i is rainfall intensity ($mm\ h^{-1}$), and A is area of the field or watershed (km^2), and 3.6 is a constant for conversion. The C indicates the amount of rainfall that becomes runoff in a single event and varies by storm event. The C values for different land use and cropping systems under the four hydrologic groups were summarized by Schwab et al. (1993). The modified rational method is expressed as

$$q = \frac{Q \times \alpha_{tc} \times A}{3.6 \times T_c} \tag{2.34}$$

where α_{tc} is runoff fraction during the time of concentration, and A is watershed area in km^2. Replacing

$$C = \frac{Q(mm)}{R_{day}(mm)} \tag{2.35}$$

$$i = \frac{R_{tc}}{T_c} \tag{2.36}$$

in Eq. (2.33) results in

$$q = \frac{Q \times R_{tc} \times A}{3.6 \times R_{day} \times T_c} \tag{2.37}$$

where

$$R_{tc} = \alpha_{tc} \times R_{day} \text{ or } \alpha_{tc} = \frac{R_{tc}}{R_{day}} \tag{2.38}$$

which gives

$$q = \frac{Q \times \alpha_{tc} \times R_{day} \times A}{3.6 \times R_{day} \times T_c} = \frac{Q \times \alpha_{tc} \times A}{3.6 \times T_c} \tag{2.39}$$

Example 3. Estimate the peak discharge rate for designing a runoff control system for a watershed of 2.95 ha if the intensity of rainfall with a 25-yr return period for 10 min is 150 mm h^{-1}. Assume runoff coefficient equal to 0.95.

$$q = CiA = 0.95 \times \frac{0.150\,\mathrm{m}}{3600\,\mathrm{s}} \times 29500\,\mathrm{m}^2 = 1.168\,\mathrm{m}^3 s^{-1}$$

Example 4. Compute the peak runoff rate for Example 2 using the modified rational method for 2 km^2 watershed if the rainfall intensity is 50 mm fallen in 2 h and time of concentration is 1.25 h.

$$R_{tc} = i \times t_c = 25\,mm\mathrm{h}^{-1} \times 1.25\,\mathrm{h} = 31.25\,\mathrm{mm}$$
$$\alpha_{tc} = \frac{R_{tc}}{R_{day}} = \frac{31.25\,\mathrm{mm}}{50\,\mathrm{mm}} = 0.625$$
$$q = \frac{Q \times \alpha_{tc} \times A}{3.6 \times T_c} = \frac{7.17\,\mathrm{mm} \times 0.625 \times 2\,\mathrm{km}^2}{3.6 \times 1.25\,\mathrm{h}} = \frac{8.963}{4.5} = 1.99\,\mathrm{m}^3 s^{-1}$$

2.7 Soil Properties Affecting Erodibility

Erodibility is the soil's susceptibility to erosion. It is a dynamic attribute that changes over time and space with soil properties. Field, plot, and lab studies are used to assess soil erodibility. Erosion indexes have often been used to estimate the soil erodibility. Soil texture, soil structure (e.g., macroporosity, aggregate properties), organic matter content, hydraulic properties, and wettability are some of the factors which affect erodibility.

2.7.1 Texture

Sandy soils are less cohesive than clayey soils and thus aggregates with high sand content are more easily detached. While a well-aggregated clayey soil is more resistant to erosion than coarse-textured soils, once detached, the clay particles are readily removed by runoff due to their smaller size. Silty soils derived from loess parent material are the most erodible type of soil. Water infiltration is positively correlated with an increase in coarse soil particles and negatively with an increase in fine particles (Wuest et al., 2006). Sandy soils have larger macropores and absorb water more rapidly than clayey soils. Macropores conduct water more rapidly than micropores. Under low intensity rains, sandy soils produce less runoff than clayey soils. Most of the rain falling on clayey soils is partitioned into runoff due to the abundant micropores which reduce water infiltration. While sandy soils have lower total porosity than clayey soils, their porosity consists mostly of macropores.

2.7.2 Structure

Soil structure, architectural arrangement of soil particles, confines pore space, biological entities, and aggregates of different size, shape and stability. The soil's ability to resist erosion depends on its structure. Soils with poor soil structure are more detachable, unstable, and susceptible to compaction, thereby have low water infiltration and high runoff rates. Because soil structure is a qualitative term, related parameters such as water infiltration, air permeability, and soil organic matter dynamics are used as indicators of soil structural development. Assessment of aggregate structural properties is also a useful approach provided that soil structural stability at the aggregate level determines the macroscale structural attributes of the whole soil to withstand erosion.

Various techniques exist for characterizing and modeling soil structure. Advanced techniques of soil structure modeling are designed to capture the heterogeneity of soil structure and relate these quantifications to various processes (e.g., erosion). Techniques focusing on the whole soil combined with aggregate characterization may provide more insights into the soil structure dynamics. Among the current techniques are tomography, neural networks, and fractals (Young et al., 2001). Tomography allows the investigation of the interior architectural design of soil and permits the 3D visualization of soil structure. By using this approach, it is possible to examine the geometry and distribution of macropore and micropore networks within the soil, which contribute to air and water flow. The use of neural networks is another approach to look at the soil structural attributes for retaining water, storing organic matter, and resisting erosion. Soil fragmentation during tillage and its susceptibility to soil erosion are governed by the fractal theory. This theory involves the study of the complexity of soil particle arrangement, tortuosity, and abundance of soil pores, which are essential to explain processes of water flow through the soil. These relatively new techniques can help to quantify soil structural attributes.

2.7.3 Surface Sealing

Surface sealing is a major cause of low water infiltration rate, and high risks of runoff and soil erosion. Surface sealing results from the combined effect of raindrop impact on soil surface and deflocculation of clay particles. Initially, the rainfall impact breaks exposed surfaces of soil aggregates, and disperses clay creating a thin and compact layer of slaked fine particles at the soil surface, known as surface seals. The settled fine particles fill and clog the water conducting soil pores significantly decreasing the infiltration rate and increasing surface runoff and soil transport. The process of formation of surface seals is complex and depends on the rainfall amount, intensity, runoff rate, soil surface conditions (e.g., residue mulch), soil textural class, vegetative cover, and tillage management. When dry, surface seals form crusts with a thickness ranging between 0.1 and 5 cm.

2.7.4 Aggregate Properties

The adherence of soil primary particles to each other more strongly than to the neighboring soil particles creates an aggregate. Aggregate attributes are important to understanding and modeling soil erosional processes particularly in well-aggregated soils. Soil properties in relation to stability and erodibility are often assessed using large samples rather than structural units or discrete aggregates. As yet, attributes of macro- and micro-aggregates determine the rates of soil detachment by rainfall and runoff. Aggregate structural properties such as stability, strength, density, sorptivity, and wettability affect soil erodibility.

2.7.4.1 Stability

Stability refers to the ability of an aggregate to withstand the destructive applied forces (e.g., raindrops). It is a function of the cohesive forces that hold the primary particles together. Soil detachment by rainfall depends on the ability of surface aggregates to resist the disruptive energy of raindrops. Raindrop energy must overcome the cohesive energy of the aggregate to disintegrate it. Wet-sieving which involves submergence and oscillation of a group of aggregates is a common lab technique to assess aggregate stability. This method uses a group of aggregates rather than a single aggregate. Tests of aggregate stability on individual aggregates using simulated raindrop technique account for the heterogeneity of field aggregates and provide additional insights into aggregate dynamics in relation to soil erosion.

Aggregate stability is a function of soil texture, soil organic matter content, cation exchange capacity (CEC), presence of cementing agents, tillage and cropping systems, manure application, and residue management. Aggregates from plowed soils are structurally unstable and are dispersed readily by raindrop energies unlike those from undisturbed agricultural systems (e.g., pasture, no-till). Intensive tillage interrupts the natural soil structural development and causes the breakdown of stable aggregates and loss of soil organic matter. Abundant surface residue cover in interaction with reduced soil disturbance results in stable aggregates. The kinetic energy required to disintegrate aggregates increases with increase in size of stable aggregates. Thus, large and stable aggregates are less erodible than small and weak aggregates. Small aggregates are also easily transported in runoff and contribute to higher soil losses. The homogenization and seasonal mixing of the plow layer in tilled soils form weak aggregates, which are easily detached by rain regardless of size. Macro- and micro-aggregates in undisturbed soils are stable and have slow turnover rates due to their high soil organic matter content.

2.7.4.2 Strength

Aggregate strength is a dynamic property that affects soil erodibility. One of the most useful mechanical properties of aggregates is tensile strength, which refers to the force required to break an aggregate. It is a measure of the inter- and intra-aggregate bonding forces and the amount of soil aggregation. Depending on the soil and management, air- dry aggregates from plowed soils following reconsolidation tend to have higher tensile strength than those from no-till soils. The higher tensile strength does not, however, always translate into higher aggregate stability because, during wet-sieving, air-dry aggregates from plowed soils slake rapidly in spite of their high air-dry strength. This is attributed to the fact that plowed soils have lower organic matter content compared to no-till soils, which have more organic binding agents to form stable aggregates. Blanco-Canqui and Lal (2008a) observed that corn stover removal from no-till soils reduced tensile strength of aggregates due to the decrease in soil organic matter content by stover removal.

2.7.4.3 Density

Compacted soils often have low number of macropores, high bulk density, and low water infiltration and high runoff rates. Tillage and residue management and manure application affect aggregate density. Because of the rapid post-tillage consolidation in concomitance with the low soil organic matter content, plowed soils generally have higher aggregate density and lower number of macropores than no-till soils. Increased soil organic matter content and bioturbation in no-till dilute the aggregate density and increase soil macroporosity, which is important to increasing water infiltration rate and reducing runoff rates.

2.7.4.4 Wettability

Wettability is the ability of a soil to absorb water. Some soil aggregates exhibit slight water repellency due to the coating of their surface by soil organic matter -derived exudates and humic substances which form hydrophobic surface films (Chenu et al., 2000). Moderate water repellency is beneficial to soil structural stability because it reduces slaking and increases stability of aggregates, but high water repellency can significantly reduce water infiltration and increase runoff rates. Quantity and quality of soil organic matter influence hydrophobicity of aggregates. Mulching and manure application induce some degree of water repellency by increasing soil organic matter content. Soil aggregates under no-till tend to have higher water repellency than those under plow tillage (Blanco-Canqui and Lal, 2008b) (Fig. 2.5). Crop residue removal reduces the water repellency in no-till soils due to the reduction in soil organic matter content (Blanco-Canqui and Lal, 2008a). Techniques for assessing water repellency include water drop penetration time test, the critical surface tension test, water repellency index, and the contact angle method.

2.7.5 Antecedent Soil Water Content

The antecedent water content influences the rate of soil detachment. The wetter the soil, the less the pore space available for rainwater absorption, the greater the runoff and soil erosion. The role of initial water content on detachment and soil erosion is influenced by rainfall characteristics, soil texture, and soil organic matter content. Influence of antecedent soil water content on runoff is relatively small in compacted

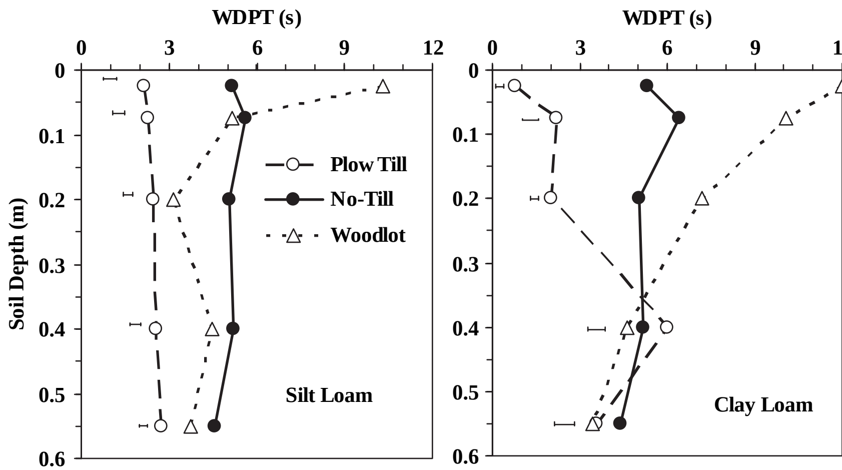

Fig. 2.5 No-till practices can increase water drop penetration time (WDPT) or induce slight water repellency to soil due to increases in soil organic matter content (After Blanco-Canqui and Lal, 2008b). Error bars at each depth interval are the LSD values ($P < 0.05$)

soils or when the rain is intense. The kinetic energy of rain required to break soil aggregates decreases with decrease in soil water content. Air-dry aggregates are more dispersible than moist aggregates because rapid wetting of dry aggregates causes sudden release of heat of wetting and entrapped air, resulting in faster disintegration in contrast with moist aggregates.

2.7.6 Soil Organic Matter Content

The soil organic matter is one of the key factors that control the stability of aggregates. It physically, chemically, and biologically binds primary particles into aggregates. Organic materials supply cementing and binding agents and promote microbial processes responsible for the enmeshment of soil particles into stable aggregates. It is important to understand the types of organic binding agents that intervene in soil aggregation. The nature, size, stability, and configuration of aggregates depend on the action of soil organic matter -derived stabilizing agents. These organic binding agents are classified in temporary, transient, and persistent agents (Tisdall and Oades, 1982). Temporary agents consist of plant roots, mucilages, mycorrhizal hyphae, bacterial cells, and algae. These agents enmesh the mineral particles and are mainly associated with macroaggregation. Transient agents consist mainly of polysaccharides and organic mucilages resulting from microbial processes of plant and animal tissues and exudations. Persistent agents include highly decomposed organic materials such as humic compounds, polymers, and polyvalent cations and are associated with microaggregate dynamics. These compounds are found inside microaggregates forming clay-humic complexes and chelates.

The stability of soil aggregates increases with increase in organic matter content. Plant roots, residue mulching, and manure addition are the main sources of organic matter and have beneficial impacts on improving aggregate stability. Stable aggregates require a higher rainfall kinetic energy to be disintegrated. The high macroporosity and permeability of these aggregates decrease runoff and soil erosion rates. Minimizing soil disturbance is a strategy to reduce organic matter oxidation and stabilize the soil structure.

2.7.7 Water Transmission Properties

2.7.7.1 Water Infiltration

Runoff occurs when the rate of applied surface water from rain or irrigation exceeds the water infiltration capacity of the soil. At the beginning of a rain event, most of the rain is absorbed by the soil, but as the soil becomes saturated, a portion of rain fills the surface depressions, and the excess water runs off the field. The amount of water infiltrated during a rainfall event determines the amount of water lost as runoff. Water from rain or irrigation infiltrates into the soil under the influence of matric and gravitational forces. During infiltration, the soil layers becomes wetter over time as the wetting front advances into layers of lower water content as compared to overlying soil.

2.7.7.2 Prediction of Water Infiltration

A number of models are available for predicting water infiltration and estimating runoff rate for a rainfall event. The fundamental basis for understanding vertical infiltration is the Richard's equation expressed as:

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial z}\left(D_w(\theta)\frac{\partial \theta}{\partial z}\right) + \frac{\partial K(\theta)}{\partial \theta} \tag{2.40}$$

where θ is water content, t is time, D_w is water diffusivity function, and $K(\theta)$ is unsaturated hydraulic conductivity in the z flow direction. Eq. (2.40) represents a process-based and nonlinear model and it can not be solved analytically. Philip (1957) developed a simplified form of flow equation where cumulative infiltration (I) and infiltration rate (i) are estimated as

$$I = St^{\frac{1}{2}} + At \tag{2.41}$$

$$\frac{dI}{dt} = i = \frac{1}{2}St^{-\frac{1}{2}} + A \tag{2.42}$$

where S is sorptivity as a function of initial and final soil water content, and A is saturated hydraulic conductivity which is nearly equal to the constant infiltration rate.

One of the earliest infiltration models was developed by Green and Ampt (1911), which in is its simplest form is expressed as:

$$i = i_c + \frac{b}{I} \tag{2.43}$$

where i_c is steady infiltration rate, b is a constant. The Green–Ampt model is a process-based model and is widely used to estimate water infiltration and determine the exact time when and how much runoff occurs during a rainfall event.

Example 5. Estimate the infiltration rate for a cumulative infiltration of 300 mm if the constant infiltration for this particular soil is 6 mm h^{-1}. The infiltration rate was 18 mm h^{-1} at a cumulative infiltration of 90 mm.

$$i = i_c + \frac{b}{I} = 6\,mm\,h^{-1} + \frac{b}{90\,mm\,h^{-1}} = 18\,mm\,h^{-1}$$
$$b = 1080\,mm\,h^{-1}$$
$$i = i_c + \frac{b}{I} = 6\,mm\,h^{-1} + \frac{1080\,mm^2\,h^{-1}}{300\,mm\,h^{-1}} = 9.6\,mm\,h^{-1}$$

Example 6. How does the rainfall intensity affect the total cumulative water infiltration in the soil of Example 1 if the intensity changes from 1 cm h^{-1} to 4 cm h^{-1}?

$$I = \frac{b}{(i - i_c)} = \frac{1080\,mm^2\,h^{-1}}{(10 - 6)\,mm\,h^{-1}} = 270\,mm$$
$$I = \frac{b}{(i - i_c)} = \frac{1080\,mm^2\,h^{-1}}{(40 - 6)\,mm\,h^{-1}} = 32\,mm$$

It is clear from this example that cumulative water infiltration decreases rapidly with increase in rainfall intensity due to surface sealing of pores and soil dispersion. The higher the rainfall rate, the lower the amount of water that can infiltrate into the soil without exceeding the infiltration capacity of the soil and greater the chances for runoff occurrence.

Example 7. How much runoff would occur from the soil in Example 6 if rain fell at 4 cm h^{-1} for 2 h assuming that surface water storage is 1 cm and the evaporation rate is 0.25 cm h^{-1}? How about if rain fell at 1 cm h^{-1} for 2 h?

$$\begin{aligned} \text{Total amount of rainfall} &= 8\text{ cm} = 80\text{ mm} \\ \text{Runoff amount} &= \text{Rainfall} - (\text{Infiltration} + \text{Surface Storage} \\ &\quad + \text{Evaporation}) \\ &= 80 - (32\text{ mm} + 10\text{ mm} + 2.5\text{ mm}) = 35.5\text{ mm} \end{aligned}$$

No runoff and soil erosion would occur from the soil receiving rain at an intensity of 1 cm h^{-1} as the cumulative infiltration is greater than the rainfall rate.

2.7.7.3 Saturated Hydraulic Conductivity

Saturated hydraulic conductivity (K_{sat}) defined as the ability of a soil to conduct water under saturated conditions is an essential parameter that affects soil hydrology and thereby erodibility. It influences runoff, drainage, water infiltration, and leaching. The K_{sat} (mm h^{-1}) is calculated using the Darcy's law:

$$q_s = -K_s \frac{\alpha H}{\alpha z} = -K_s \frac{(H_2 - H_1)}{(z_2 - z_1)} \tag{2.44}$$

where q_s is water flux (mm h^{-1}), H_1 is hydraulic head at soil point z_1 (top) (mm) and H_2 is the head at z_2 (mm). Soil texture and macroporosity are the main parameters that affect K_{sat}. Clay soils typically have low K_{sat} values while sandy soils have high values. For example, claypan soils (Alfisols) in the midwest USA covering about 4 Mha can have K_{sat} as low as 1.83 µm h^{-1} because of the presence of an argillic horizon 130–460 mm deep, with clay contents >450 g kg^{-1} (Jamison et al., 1968; Blanco-Canqui et al., 2002). These claypan soils may perch water and create lateral flow or interflow during springtime when soils remain practically saturated. Runoff rates may be equal to rainfall on clayey soils under saturated conditions. The subsurface horizons of low K_{sat} underlying layers of high K_{sat} control the saturated water flow. Blanco-Canqui et al. (2002) reported that K_{sat} of surface 0–30 cm soil depth was 71 mm h^{-1} while that of the underlying layers was only 1.83 µm h^{-1}. Evaluation of K_{sat} for the whole soil profile is necessary for explaining the hydrology of soils for accurate soil erosion and runoff characterization.

Runoff predictions are sensitive to the initial K_{sat} values (Blanco-Canqui et al., 2002). For example, the Water Erosion Prediction Project (WEPP) uses effective K_{sat} to predict runoff (Flanagan and Nearing, 1995). Because measured values are not always available for all soils, WEPP estimates effective K_{sat} based on approximate relationships between soil properties and runoff data for various soil types (Zhang et al., 1995).

Alternatively, the effective K_{sat} (K_{eff}) for the whole soil profile based on measured values can be calculated as (Jury et al., 1991) follows:

$$K_{eff} = \frac{\sum_{J=1}^{N} L_j}{\sum_{J=1}^{N} \left(\frac{L_J}{K_J}\right)} = \frac{L_T}{\left(\frac{L_1}{K_1} + \frac{L_2}{K_2} + \cdots\cdots\cdots + \frac{L_N}{K_N}\right)} \tag{2.45}$$

where L_j is thickness of each soil layer (cm), L_T is total thickness for the depth of interest (cm), and K_j is measured K_{sat} for each soil layer. The K_{eff} varies among soils depending on the layering and depth of soil profile. The best approach to estimate K_{eff} would be to evaluate soil properties by depth for each soil although this may be too costly and time-consuming for routine use. The high variability in input K_{sat} has the undesirable effect of producing inaccurate runoff predictions. Measurement of K_{sat} under in situ conditions rather than on small cores is advisable to better portray the macropore structure and eliminate preferential flow, called by-pass flow.

2.8 Measuring Erosion

Data on the amount of soil transported from a field are required to:

- assess the magnitude or severity of erosion and its effects on soil productivity,
- develop mathematical models and test their applicability for soil erosion prediction,
- design and establish erosion control practices,
- understand and manage sedimentation in depositional areas, and
- ascertain effects of erosion on water pollution.

Data on soil erosion rates have been traditionally obtained using laboratory and field plot experiments under natural and simulated rainfall conditions. Various types of laboratory-scale and field-scale rainfall simulators are used to simulate soil erosion (Fig. 2.6). Measuring soil erosion from plots requires the consideration of plot size and knowledge of factors that affect data variability. Differences in the amount of soil erosion from two identical plots under the same soil, management, and climate conditions illustrate natural variability, which is not due to human or experimental error. Choice of the plot size and proper replication are ways to minimize the measurement variability.

There are three types of erosion plots: micro, medium or USLE plots, large plots or watersheds. The amount of soil lost per unit area varies depending on the plot size. On a unit area basis, large plots often register higher soil erosion as compared to

Fig. 2.6 The Swanson type rotating boom rainfall simulator (Photo by H. Blanco). The simulator booms are equipped with nozzles positioned at radii of 1.5, 3.0, 4.5, 6.0, and 7.6 m. Booms and nozzles rotate in a circle, and the wetted diameter is about 16 m

micro plots. Large plots captures interrill, rill, and possibly ephemeral gully erosion are preferable over micro plots (Bagarello and Ferro, 2004). Choice of plot size and measurement approach depends on the purpose of the study and the erosion phenomena (interrill, rill, and gully erosion) under interest.

Micro plots. The size of small plots can vary from 0.05 to about 2 m^2. These microplots are frequently used in laboratory experiments under simulated rainfall conditions to provide hands-on opportunity to manipulate and understand principles of soil erosion processes and factors. Micro plots allow the isolation of a specific or part of an erosion process for a detailed study of physics of erosion under controlled conditions. Micro plots are particularly suitable for studying interrill erosion. Stability, disintegration, and wettability of aggregate and surface sealing are some of the processes studied in the lab.

Medium or USLE plots. The size of the medium plots is often similar to the size of the standard plots (4×22.1 m) used for the validation of the USLE model. Many have used the medium plots to collect erosion data and validate the USLE for local conditions. The minimum width should be at least 2 m in order to minimize the effect of plot boundary influence on soil erosion.

Large plots or watersheds. The size of large plots is at least 100 m^2 and is suitable for studying combined processes of rill and interrill erosion. Large plots portray the erosion occurring at large field scale conditions and are used to test one or various hypotheses of the effects of different management scenarios simulating typical local and regional practices. These plots represent a sample of the landscape and capture the different erosional phases. Watersheds equipped with runoff sampling devices are the ideal choice for assessing rill and even ephemeral gully erosion. The long-term (>30 yr) and large (>1 ha) cultivated watersheds at the North Appalachian Experimental Watersheds in Coshocton, OH equipped with complete runoff and soil loss monitoring structure for continuous runoff sampling are an illustration of large plots (Shipitalo and Edwards, 1998). Watershed studies permit comparisons of data with those from small plots.

Summary

Water erosion is the principal component of total soil erosion. Runoff is the main driver of water erosion. While erosion is a vital process of soil formation, accelerated erosion adversely affects soil and environmental quality. The main types of water erosion are: splash, interrill, rill, gully, streambank, and tunnel erosion. Understanding the processes and factors of water erosion is critical to manage and develop erosion control practices. The water erosion process starts with detachment of soil aggregates under raindrop impacts followed by transport of detached particles and deposition of soil particles. Detachment of soil particles causes surface sealing, thereby reducing water infiltration and causing runoff and soil loss. Climate, vegetative cover, topography, and soil properties are predominant factors that affect water erosion. Surface cover consisting of growing vegetation or residue mulch is a natural defense against erosive forces of rain. It intercepts and reduces the erosive energy of raindrops, slows runoff velocity, filters soil particles in runoff, improves soil properties, and reduces soil erodibility. Amount, intensity, terminal, and drop size control the rainfall energy.

The runoff volume is normally computed using the runoff curve number method, which is based on soil properties, antecedent water content, and vegetative cover. Impervious areas (e.g., paved surfaces, compacted soils) generate larger amounts of runoff than pervious areas with vegetative cover surface and rough surface conditions. The maximum rate of runoff from a rainfall event is estimated based on the rainfall intensity and area of the field. Soil erodibility, the soil's susceptibility to erosion, is determined by soil texture, macroporosity, aggregate stability, organic matter content, hydraulic properties, wettability, and other properties. Determining the amount of runoff through direct measurement and modeling is important to designing and establishing erosion control practices, and managing sedimentation and water pollution. Microplots, medium or USLE plots, and large plots are used for collecting runoff and studying processes of rill and interrill erosion. Large or watershed plots are preferred over small plots to capture variability of the effects of different management scenarios on water erosion.

Study Questions

1. Compute the infiltration rate for a cumulative infiltration of 100 and 500 mm if the constant infiltration of the soil is 4 mm h^{-1}. The infiltration rate was 22 mm h^{-1} at a cumulative infiltration of 80 mm.
2. Estimate the time of concentration for a 1.5 km^2 watershed with soil hydrologic group C that has an overland slope length of 90 m with a slope of 4.5%. The channel length is 4 km with a slope of 0.7%. The Manning's coefficient of roughness for the watershed is 0.17 and that for the channel is 0.011.
3. Compute runoff depth and volume for Prob. 1 if an average precipitation of 50 mm per day fell in 2 h period. A third of the watershed consists of agricultural lands with crops planted in straight rows with 50% under good condition and 50% under poor condition. A third of the watershed consists of residential area with 40% of impervious surface. The rest of the watershed is under grazed and unburned woods with some litter cover.
4. Compute the peak runoff rate for Prob. 2 and 3 using the modified rational method.
5. Compute the peak runoff rate for Prob. 2 and 3 if the total amount of rain had fallen in A) 50 min and B) 3 h.
6. Repeat Prob. 2, 3, and 4 if the watershed had all been converted to either A) residential urban area with 70% of impervious surface (hydrologic soil group D) or B) wooded area without grazing and burning (hydrologic soil group A).
7. Discuss the types of erosion plots.
8. Explain the impact of saturated hydraulic conductivity on runoff volume. Indicate the erosion models that use this hydraulic parameter as an input parameter for predicting runoff rates.

9. Discuss different types of rainfall simulators.
10. Describe factors affecting soil erodibility.

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Chapter 3
Wind Erosion

Wind erosion, also known as eolian erosion, is a dynamic process by which soil particles are detached and displaced by the erosive forces of the wind. Wind erosion occurs when the force of wind exceeds the threshold level of soil's resistance to erosion. Geological, anthropogenic, and climatic processes control the rate and magnitude of wind erosion (Fig. 3.1). Abrupt fluctuations in weather patterns trigger severe wind storms. Wind erosion is the result of complex interactions among wind intensity, precipitation, surface roughness, soil texture and aggregation, agricultural activities, vegetation cover, and field size. Plowed soils with low organic matter content and those intensively grazed and trampled upon are the most susceptible to erosion. About 50% of the dust clouds result from deforestation and agricultural activities (Gomes et al., 2003).

3.1 Processes

Wind detaches and transports soil particles. Transported particles are deposited at some distance from the source as a result of an abrupt change in wind carrying capacity. The three dominant processes of wind erosion, similar to those of water erosion, are: *detachment, transport, and deposition* (Fig. 3.1). The mechanics and modes of soil particle movement are complex. Deposition of suspended particles depends on their size and follows the Stoke's Law. Large particles settle down first followed by particles of decreasing size. Smaller particles remain suspended forming the atmospheric dust.

The three pathways of particle transport are *suspension, saltation, and surface creep* (Fig. 3.3). The mode of transport of soil particles during wind erosion is governed by the particle size. Small particles (<0.1 mm) from pulverized soils are preferentially transported in suspension, medium-sized particles (0.1–0.5 mm) in saltation, and large particles (0.5–2 mm) by surface creeping. Because of abrasion, rebounding, and rebouncing effects, saltating and creeping particles can be broken into smaller particles and be transported in suspension. Saltation, suspension, and surface creep are not separate but interactive and simultaneous processes of transport (Fig. 3.3). The size of moving particle with wind decreases with increase in height above the soil surface (Fig. 3.4).

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Fig. 3.1 Wind erosion creates sand dunes in arid regions (Photo by H. Blanco)

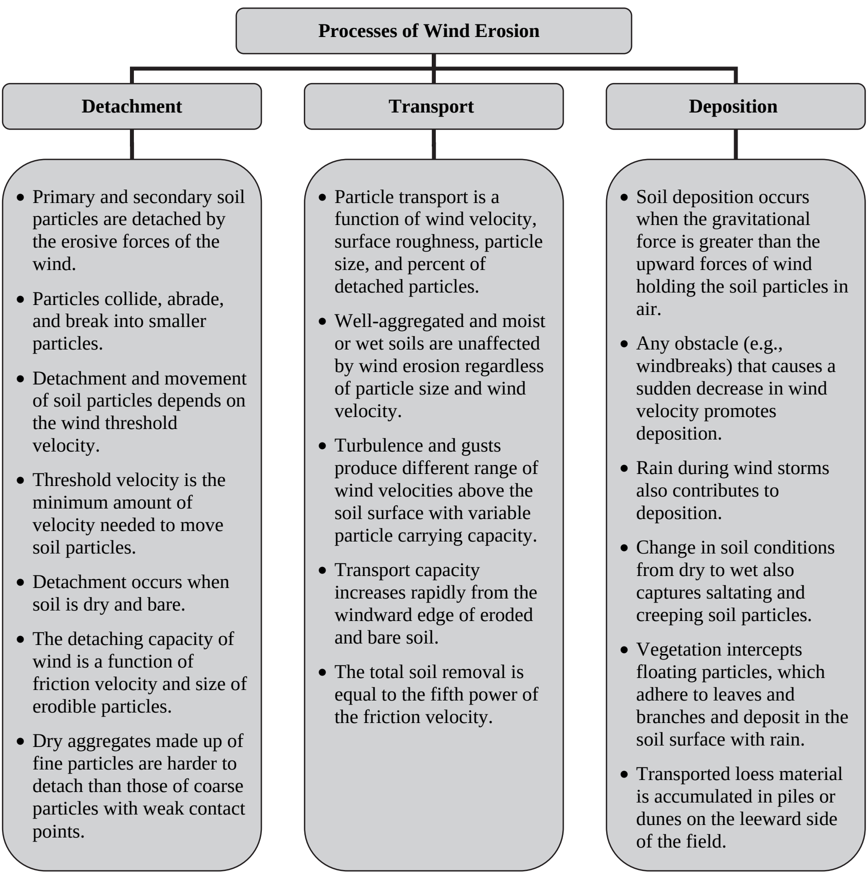


Fig. 3.2 Three main processes of wind erosion

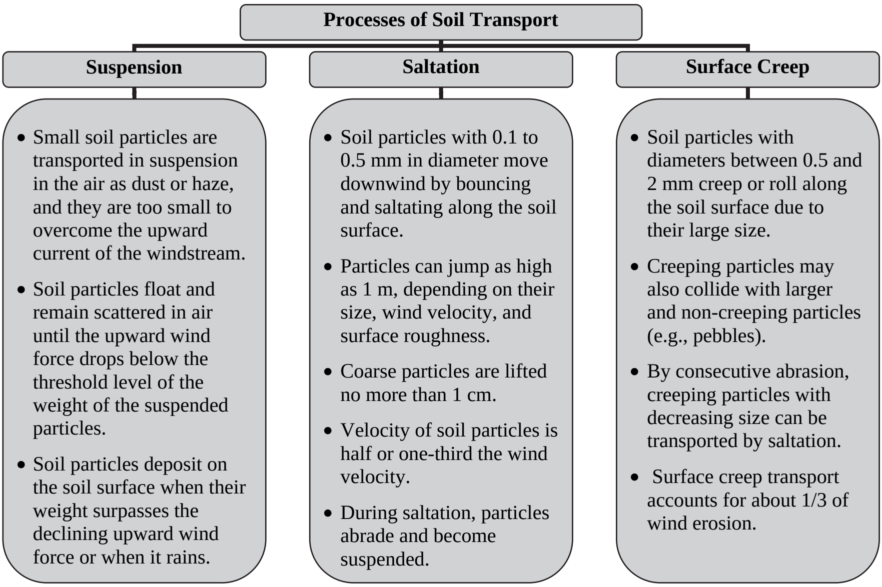


Fig. 3.3 Processes of soil transport during wind erosion

Most soil particles are transported by saltation, which represents about 50–70% of total wind erosion. About 30–40% of particles are transported by suspension while about 5–25% by surface creep (White, 1997). Saltating particles consist of primary and secondary particles carrying fine organic and inorganic particles. Travel distance of particles in suspension differs largely from that in saltation and creep. Saltating and creeping particles advance shorter distances than suspended particles (Fig. 3.4). The amount of particles transported by suspension increases with an increase in bare field area and wind velocity. Intensive wind erosion creates distinctive features. Polishing or weathering of wind-exposed sedimentary rocks (e.g., rock outcrops) is typical in areas affected by wind erosion. Large concentration of windstreams along depressions carves pits and channels, forming deflation hollows.

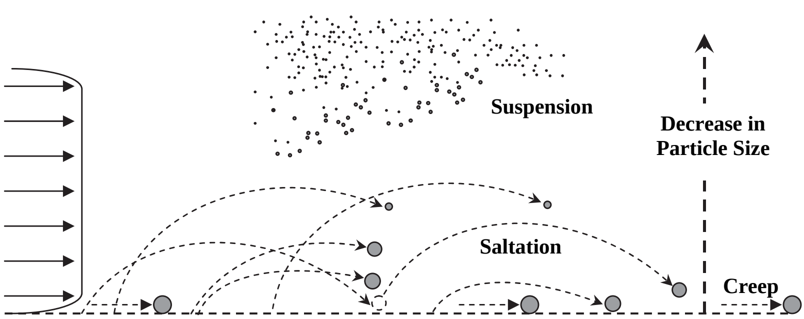


Fig. 3.4 Modes of soil particle transport by wind during erosion

The continued removal of small particles by wind leaves landscapes paved with exposed stones and pebbles in arid regions.

3.2 Factors

Wind erosion is the result of a combination of many factors associated with climate, soil, land surface, and management conditions (Table 3.1). Wind velocity, soil surface water content, surface vegetative cover, surface roughness (e.g., ridge height), aggregate stability, field length, rock volume fraction, and soil texture are the most sensitive parameters influencing wind erosion (Feng and Sharratt, 2005). There are two opposing forces that take place during soil erosion (Fig. 3.5). The force of wind which tends to move everything away faces an opposing front, which

Table 3.1 Four interactive factors affecting wind erosion dynamics

Climate	Land Surface Properties	Soil Properties	Land Use and Management
• Wind speed, duration, direction, and turbulence • Wind shear velocity • Precipitation and temperature • Radiation and evaporation • Air humidity, viscosity, and pressure • Freezing and thawing	• Field slope • Length, width, and orientation of the field • Terrain roughness • Non-erodible materials (e.g., rocks, stones) • Residue orientation (e.g., flat, standing)	• Particle size distribution and particle density • Aggregate size distribution • Aggregate stability, strength, and density • Water content • Bulk density and crusting • Soil organic matter content • $CaCO_3$ concentration	• Residue management • Type of land use (e.g, forest, rangeland, and pasture) • Type of cultivation (e.g., no-till, plow till, rotations) • Fallow or bare soil • Afforestation or windbreaks

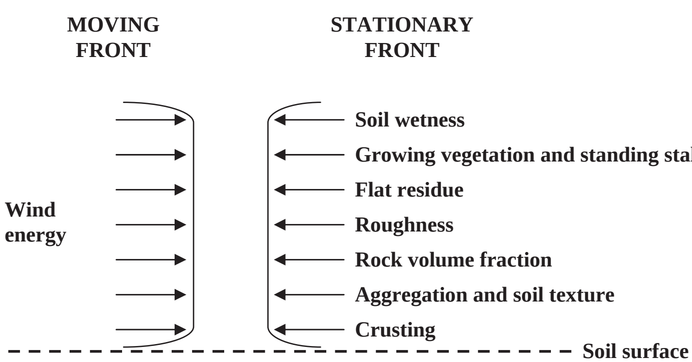


Fig. 3.5 Forces defining the rate of wind erosion (After Fryrear et al., 1998)

is the natural resistance of the soil that offsets the wind energy until the threshold level of resistance is overcome by the wind force at which point erosion is set in motion. For example, high winds increases soil transport, whereas well-aggregated soils decrease the availability of loose particles for erosion. The net effect of the opposing forces determines the rate of soil erosion. The wind is a moving force whereas the forces of soil resistance are stationary.

3.3 Wind Erosivity

Wind erosivity refers to the capacity of wind to cause soil erosion. Wind in interaction with precipitation and air temperature is the driving force of wind erosion. Wind is dynamic and composed by eddies that change rapidly in intensity and direction. Amount of rainfall and temperature fluctuations determine rates of evaporation. Measurement of wind characteristics (e.g., friction velocity, aerodynamic roughness) is often done with a portable anemometer tower. Data on wind velocity and direction, air temperature, solar radiation, relative humidity, rain amount, soil temperature and water content are essential meteorological input parameters for characterizing wind erosion.

The wind velocity must be near $8\,\mathrm{m\,s^{-1}}$ at 2 m above the soil surface for the soil particles to be displaced by wind. Fast winds cause more erosion than slow winds. Wind velocity changes on an hourly, daily, and seasonal basis. The air movement near the soil surface is small because of the drag force between air and soil surface. The drag force increases with increase in surface roughness. A rough surface changes the wind profile. Wind blowing over a flat and smooth surface is shifted to a new level when it reaches a rough surface. Wind velocity increases with height above the soil surface due to decrease in drag forces.

The wind velocity at any given distance above the soil surface with crop canopy cover is computed using the semi-logarithmic model derived from the first momentum of eddy function as

$$U_{(z)} = \frac{\mu^*}{k} \ln\left[\frac{z-d}{z_0}\right] = \frac{\mu^*}{0.4} \ln\left[\frac{z-d}{z_0}\right] \tag{3.1}$$

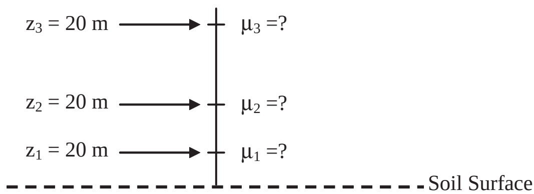
where $U_{(z)}$ is the wind velocity at height z, μ^* is the friction velocity, k is the von Karman constant equal to 0.4, d is the aerodynamic displacement height equal to 0.7 $\times$ height of roughness element, and z_0 is the aerodynamic roughness parameter assumed to be equal to 0.13 $\times$ height of roughness element (Tanner and Pelton, 1960) or 0.15 $\times$ height of roughness element (Böhner et al., 2003). Wind velocity within the canopy cover is estimated (Landsberg and James, 1971) as

$$U_{(z)} = U_h\left[1 + \alpha\left(1 - \frac{z}{h}\right)\right]^{-2} \tag{3.2}$$

$$\alpha = \left(2 \times \left(1 - \frac{d}{h}\right) \times \ln\left[\left(1 - \frac{d}{h}\right)\left(\frac{z_0}{h}\right)^{-1}\right]\right)^{-1} \tag{3.3}$$

where U_h is the wind velocity within the crop canopy (m s^{-1}), h is the canopy height (m), and α is the damping effect of the crop canopy.

Example 1. Determine the wind velocity at 10 and 20 m above the soil surface if the wind velocity at the 5 m height is 4 m s^{-1} for a field with crop height of 2 m. Assume z_0 is equal to 0.15 the height of crops.



Since d and z_0 remain the same with height above the surface, the wind velocity is estimated as per Eq. (3.1)

$$\mu_2 = \mu_1 \frac{\ln\left(\frac{z_2 - d}{z_0}\right)}{\ln\left(\frac{z_1 - d}{z_0}\right)} = 4\frac{\ln\left(\frac{10-1.4}{0.3}\right)}{\ln\left(\frac{5-1.4}{0.3}\right)} = 5.40\,\mathrm{m\,s^{-1}}$$

$$\mu_3 = \mu_2 \frac{\ln\left(\frac{z_2 - d}{z_0}\right)}{\ln\left(\frac{z_1 - d}{z_0}\right)} = 5.40\frac{\ln\left(\frac{20-1.4}{0.3}\right)}{\ln\left(\frac{10-1.4}{0.3}\right)} = 6.64\,\mathrm{m\,s^{-1}}$$

The $\mu_3 > \mu_2 > \mu_1$ indicates that velocity increases with height above the soil surface.

Threshold wind velocity refers to the velocity required to entrain a soil particle. The threshold velocity required to initiate soil movement varies with soil surface and vegetative cover conditions. It increases with increase in soil particle size. Particles that are fine and loose are entrained more easily than coarse particles under the same wind velocity. A greater wind velocity is needed to break away and move particles in undisturbed and surface covered soils.

There are two types of threshold levels: static and dynamic. The *static or minimum threshold velocity* is the velocity at which the least stable soil particles are detached but are not transported. The *dynamic or impact threshold velocity* is the velocity at which the detached particles are transported (Fryrear and Bilbro, 1998). Soil erosion rates increase exponentially with increases in wind velocity (Fig. 3.6). The rate of erosion by wind is proportional to the cube of the wind velocity above the threshold level.

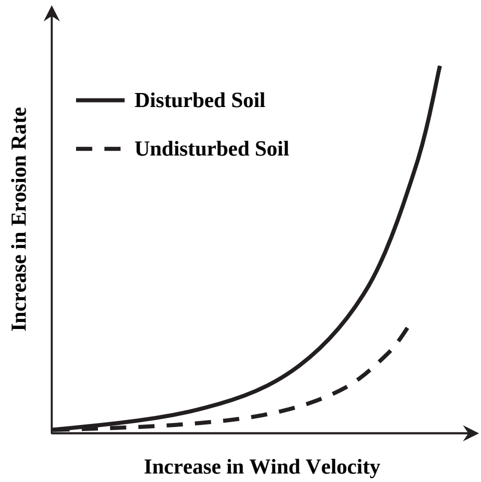


Fig. 3.6 Relationship between erosion rates and wind velocity. Erosion rates are directly proportional to the amount of exposed and loose erodible material, which is influenced by the level of soil disturbance, crusting, management, and soil texture

3.4 Soil Erodibility

Magnitude of wind erosion is a function of soil erodibility, which refers to the ability of the surface soil to resist the erosive forces of wind. Intrinsic soil properties such as texture, structure, and water content in interaction with surface roughness and living and dead vegetative cover define the rate at which the soil is detached and eroded. Any soil that is dry and loose with bare and flat surface is susceptible to wind erosion. Dry loose soil material <0.84 mm in diameter occurring on the soil surface, known as loose erodible material, is the fraction that is readily transported by wind (Zobeck, 1991). Portable vacuum devices are used to determine the amount of erodible material under field conditions (Zobeck, 1991). Soil erodibility changes dynamically on a spatial and temporal basis due to tillage and residue management.

3.4.1 Texture

Soil erodibility depends on the size distribution of soil particles and their ability to form stable macro- and micro-aggregates. Soil particles coalesce and form aggregates in interaction with organic matter. Sandy loam and sandy soils with low organic matter content develop aggregates with weak bonds and are thus the most erodible. Fine textured soils, in turn, often develop stable and strong aggregates resistant to wind erosion. Any soil that is dry and pulverized is, however, susceptible to erosion. Under these conditions, particle removal is the order of: clay>silt>fine sand, decreasing with increase in particle size.

3.4.2 Crusts

The unconsolidated and loose fine soil particles in tilled soils form seals under the influence of rain, which later develop into thin crusts or skins when soil dries out. These soil skins have textural and structural properties (e.g., water, air, and heat fluxes, mechanical bonding) completely different from the soil beneath. Crusts are more dense, stable, resistant to erosion than uncrusted soils. The rate at which crusts are degraded or eroded depends on the magnitude of the abrasive forces of the wind. Crusts temporarily protect the soil beneath until crusts are either lifted or broken apart by wind past the threshold level of velocity, at which point soil under the crusts is eroded rapidly.

Crust formation and thickness vary from soil to soil as function of soil physical, biological, and chemical properties, surface roughness, vegetative cover, and raindrop impacts. They even vary within the same soil type. Presence of stones, ridges, residue mulch, and stable aggregates confines crust formation to areas between non-erodible or stable surface materials. The fraction of soil surface covered by crusts is quantified by methods similar to those used for vegetation cover characterization. While excessive crusting can impede seedling emergence and reduce water infiltration, moderate crusting reduces wind erosion. Wind erosion rates decrease exponentially and linearly with increase in percentage of crust cover. Erosion rates from crusted soils can be 5–5000 times lower than those from uncrusted soils, depending on the wind velocity (Li et al., 2004). Wind tunnel experiments are used to assess the ability of crust to withstand abrasion by sand particles.

Some simple equations developed for estimating wind erosion rates (E) for crusted soils (Li et al., 2004) are:

$$Wind\ speed = 26\,ms^{-1} \rightarrow E = 582.41 \times exp(-0.021 \times Crust) \tag{3.4}$$

$$Wind\ speed = 18\,ms^{-1} \rightarrow E = 41.898 \times exp(-0.0147 \times Crust) \tag{3.5}$$

$$Wind\ speed = 10\,ms^{-1} \rightarrow E = 3.041 \times exp(-0.0048 \times Crust) \tag{3.6}$$

where C*rust* is in %.

3.4.3 Dry Aggregate Size Distribution

Distribution of dry aggregate size fractions is an indicator of soil's susceptibility to wind erosion. It is one of the key parameters to evaluate management impacts on soil structure and model wind erosion. The soil fraction most susceptible to wind erosion comprises aggregates <0.84 mm in diameter. Specific surface area, clay content, and organic matter content are important predictors of macro- and micro-aggregation. Stable macroaggregates withstand wind erosive forces and reduce soil detachment. A number of approaches including log-normal fractal and Weibull distributions have been used to evaluate the temporal variability of dry aggregate-size distributions in wind-erosion affected soils (Zobeck et al., 2003a).

3.4.4 Aggregate Stability

Aggregate stability and strength are directly affected by climate and soil management. Climatic factors such as amount of precipitation, freezing–thawing, wetting–drying, and freezing-drying, and management factors such as tillage, cropping, and residue management systems determine aggregate formation and stability. For example, soils remaining covered with snow or crop residue mulch have more stable aggregates, whereas those subject to intense and frequent freeze–thaw cycles develop unstable and small aggregates. Changes in soil organic matter content influence aggregation and aggregate stability. The organic matter provides binding agents to soil. Variations in the amount of residue left on the soil surface induce rapid change in soil organic matter content and soil aggregation.

3.4.5 Soil Surface Roughness

Surface roughness affects evaporation rates, radiation, soil temperature, soil water storage, surface tortuosity, and saltation and rolling of soil particles. Ridges, clods, and aggregates are responsible for the increased roughness of the soil surface. Height, shape, density, and number of tillage ridges and clods determine soil surface roughness. A rougher soil surface obstructs wind flow and increases the threshold wind velocity needed to move the soil particles. Wind impacts first on the windward faces of knolls of the field. On topographically complex fields, wind is funneled through the valleys as concentrated forces. While moderate surface roughness helps with reducing wind erosion, high surface roughness causes turbulence and increases wind erosion risks (Schwab et al., 1993).

Stable, abundant, and large clods oriented perpendicular to the prevailing wind direction absorb wind energy and trap particles, and reduce soil movement. There are four classes of surface roughness: soil particles (<2 mm diam.), random roughness (10-cm diam.), tillage-induced roughness (10- and 30-cm diam.), and field topographic roughness (>30 cm) (Römkens and Wang, 1986). The roughness created by tillage and traffic form oriented and random roughness. Oriented roughness is where ridges follow a particular direction, whereas random roughness refers to random distribution of clods. Pin meters, roller chain, set of chains, and random roughness index are methods to measure soil roughness (Merrill, 1998).

Ridges are prone to rapid changes from rain, traffic, and cultivation, and thus provide temporary measures against wind erosion. Bare soil surface managed with ridge tillage but with little or no residue cover is subject to abrasion and rapid soil loss by erosion when dry. Unsheltered ridges are continuously abraded by saltating and creeping particles. Soil clods produced by tillage can compensate for the lack of residues but most of these clods are unstable, short-lived, and easily eroded unless intermixed with crop residues. Wetting and drying processes contribute to the demise of clods. Reduced tillage creates stable clods which in interaction with surface residues create a rough protective cover against wind erosion.

3.4.6 Soil Water Content

Soil water content is one of the most important determinants of wind erosion. A wet or moist surface soil is not easily eroded by wind. Wind erosion rates decrease exponentially with increase in soil water content owing to the cohesive force of water. The wind threshold level increases with soil water content following a power function. Wind erosion rate decreases rapidly with increase in soil water content. Soil wetness is short-lived in bare and sandy soils under strong winds. Vegetative and residue mulch cover enhance water storage by reducing fluctuations in temperature, evaporation, moderating heat fluxes conductivity, and reducing sublimation and pressure vapor oscillations (Layton et al., 1993). High air temperature dries the soil rapidly and increases wind erosion potential. High water content under residue mulch consolidates aggregates and prevents aggregate drying, cracking, and detachment.

3.4.7 Wind Affected Area

Large and bare fields are more susceptible to erosion than small and protected fields. The wind erosion increases with increase in unsheltered distance because large and unobstructed fields allow wind to gain momentum and its erosive energy. Bare fields with the longest axis parallel to the wind direction erode more than fields with main axis perpendicular to the wind. Fields must be oriented perpendicular to the predominant wind direction in the region. Soil movement grows with increase in length of eroding fields until a *maximum movement rate* is reached. The *maximum rate* refers to the largest concentration of soil particles that a wind can transport. On highly erodible soils, the maximum rate is reached over a short distance. The maximum rate is about 1.8 Mg per 5 m of field width per hour for a wind velocity of 17.9 m s^{-1} at 15 m height for a bare, smooth, and dry field (Chepil, 1959). The maximum rate is not reached in small fields.

3.4.8 Surface Cover

Vegetative cover is the single most important shelter against wind erosion. Plants protect the soil surface and their roots anchor the soil, improve aggregation, and decrease soil erodibility. Living or dead vegetative cover protects the soil. Establishment of permanent vegetative cover and adoption of conservation tillage including windbreaks, strip-cropping, stubble mulch tillage, no-till, and reduced tillage are effective measures to minimize wind erosion. Vegetative cover slows wind velocity and reduces soil erosion rates. The threshold wind velocity increases and soil erosion rates decrease with increase in percentage of vegetative cover. Various site-specific relationships between erosion and vegetation cover exist (Zhang et al., 2007) as follows:

$$U_z = 5.39 + 0.0638 \times \exp\left(\frac{VC}{-12.35}\right) \tag{3.7}$$

$$Q = a + b \times \exp\left(\frac{VC}{c}\right) \tag{3.8}$$

where U_z is the threshold wind velocity (m s^{-1}), VC represents the vegetation coverage (%), and Q is soil erosion rate (kg m^{-2} min^{-1}), and a, b, and c are regression coefficients, which vary with wind velocity.

3.4.9 Management-Induced Changes

Most of the major factors affecting wind erosion are also influenced by management (e.g., tillage, residue management, cropping systems) in interaction with climate. For example, intensive tillage induces rapid modifications in soil properties as it breaks, inverts, mixes, and pulverizes soil. Tillage also induces spatial and temporal changes in soil hydrologic properties (e.g., water retention). Amount of crop residue mulch affects wind erosivity and soil erodibility. Soils vary in their response to management. Undisturbed soils are more resilient and less erodible than disturbed soils.

3.5 Measuring Wind Erosion

Accurate measurement of wind erosion rates is important to assessing the magnitude of the erosion problems, developing, validating, and calibrating predictive models, and establishing erosion control practices when necessary. A number of sampling devices with varying sampling efficiencies and design characteristics exist for trapping sediment flux by wind. The choice of sediment sampler depends upon the size of windblown particles to be collected. Designing an isokinetic sampler, which means that air flow through the sampler intake is the same as that in the surrounding environment, has been a difficult task, and collection of particularly fine particles has been subject to errors at high wind velocities. The suspended finer particles commonly follow the wave of wind streamlines, while the relatively larger particles in suspension are affected by their inertia and often cross streamlines (Shao et al., 1993). Because a sampler place in the field is an obstacle to the normal wind flow, it alters the friction velocity and distorts the flow of soil particles, which often reduces the efficiency of samplers for capturing representative sediment samples.

3.5.1 Efficiency of Sediment Samplers

The efficiency of collectors varies depending on a number of factors (Goossens and Offer, 2000):

Design. The size, shape, and type of material used for the construction of samplers affect the aerodynamics of wind flow. Configuration of samplers, installation, and sampling procedures are important characteristics of efficiency.

Wind velocity. The velocity at which the wind flows inside the sampler with respect to the incident flow determines the sampler efficiency. The faster the wind velocity, the greater the difficulties for collecting representative samples.

Particle size. Coarse particles are more easily captured than fine particles due to the higher inertia of larger particles. Smaller particles flow intermixed with wind streamlines and are not affected as much by inertia as larger particles. Collecting fine particles requires the design of special traps (e.g., fine wire mesh) while ensuring an uninterrupted flow of wind through the sampler. While reducing the mesh size increases the trapping efficiency of fine particles, it greatly increases the flow distortion.

Duration of sampling. The build-up of collected particles inside the passive samplers during a wind storm event may reduce the efficiency of samplers. The saturation of the traps (fine mesh) with dust reduces the rate of air flow unless an active sampler is used with a continuous pumping system, which prevents the filter from saturation.

3.5.2 Types of Sediment Samplers

Samplers are grouped into two main categories: *active* and *passive*. The active samplers are equipped with pumping devices to maintain isokinetic conditions and are suitable for collecting $<2\,\mu$m fine particles (e.g., clay, emissions of PM10 and PM2.5), whereas the passive samplers do not use a pumping mechanism and are appropriate for collecting $>40\,\mu$m coarse particles (Shao et al., 1993). Field measurements of wind erosion often rely on passive samplers, which are less expensive and more portable than active samplers.

According to the particle size, there are sand and airborne dust samplers. Based on the location of the sampling orifices, samplers are used for collecting *horizontal* and *vertical* sediment flux. The movement and deposition of windblown particles are the net result of horizontal and vertical fluxes. Samplers are also classified as *single-point* or *depth integrating* samplers. Single-point samplers have a small inlet orifice (e.g., 10 mm wide by 20 mm high) (Shao et al., 1993), while those designed for collecting vertically integrated samples have a rectangular inlet opening (e.g., 20 mm wide by 500 mm high) (Nickling and McKenna Neuman, 1997). These samplers are covered with a wire mesh of 40 and 60 μm in the back to trap particles with <40 and <60 μm in diameter, respectively.

The first and passive depth integrating sampler was the Bagnold trap (Bagnold, 1941). This collector was modified to active sampler for use in current wind tunnel tests. The Modified Wilson and Cooke (MWAC) and the Big Spring Number Eight (BSNE) are the most popular samplers for collecting airborne dust and saltating particles (Zobeck et al., 2003b). Other samplers include the suspended sediment trap, wedge dust flux gauge, marble dust collector, high volume dust sampler, the modified Sartorious sampler, and Leach trap (Goossens and Offer, 2000). The efficiency of the samplers decreases with decrease in sediment particle size (<40 µm) unless the samplers are connected in series with a pumping device that produce low and high volume suctions to maintain isokinetic conditions.

3.5.2.1 Wind-Tunnel Method

The most common method to directly measure wind erosion is by means of wind tunnels. This method uses transparent tunnels or tubes to monitor wind flow characteristics and soil particle transport dynamics through the tunnel. While this technique is mostly used in the lab for developing and validating soil erosion models, portable units are used in the field (Pietersma et al., 1996). Large fans are used to simulate different wind intensities and sediment samples collected over time. Most of the available models of wind erosion prediction are validated against data from wind tunnel experiments.

3.5.2.2 Point Measurements

Changes in topsoil and profile thickness are rapid in soils under severe erosion. Excessive wind erosion causes visual changes in soil surface features such the exposure of stones, rocks, and plant roots (e.g., tree roots). These changes in soil level with respect to a reference point can provide estimates of wind erosion rates. Many simple techniques such as the use of erosion pins (e.g., nail, rods), paint collars, and profile meter are available for making point measurements. A large number of replicates are required to obtain credible estimates. In some soils, significant changes in soil level are detectable only after a long period of monitoring.

The pin method consists of driving pins into the soil to monitor over time changes in soil level due to erosion with respect to the nail top. Painting collars around tree trunks, shrubs, rocks, and fence posts is another technique. Decrease in soil level with reference to the paint lines gives an estimate of soil lost by wind. The profile meter is similar to the device used to determine soil roughness caused by tillage. It has two permanent vertical supports and a horizontal bar with a number of adjustable rods to measure surface roughness and soil depth change. These techniques provide only rough estimates and have limited use for understanding the dynamics of wind erosion processes.

3.5.2.3 Radionuclide Fallouts

The fallout of radionuclide ^{137}Cs from nuclear tests performed in the 1950’s and 1960’s offers an opportunity to quantify wind erosion rates over large areas (Chappell and Warren, 2003). By using the ^{137}Cs approach, the spatial distribution of ^{137}Cs is quantified and related to total soil loss by wind assuming that erosion by water and tillage are negligible. The ^{137}Cs activity (Bq kg^{-1}) is measured on soil samples by spectrometry equipped with x-ray detectors (Chappell and Warren, 2003). Soil samples are collected from wind-erosion affected areas (e.g., croplands) and from uneroded sites (control) for comparisons purposes. Models and variograms are fitted to the measured data to map the ^{137}Cs distribution across the fields of interest. The use of ^{137}Cs as a tracer of wind erosion rates is relatively new. In the Qinghai-Tibetan Plateau, the use of ^{137}Cs was found to be a sensitive technique to estimate wind erosion, which were 84 Mg ha^{-1} yr^{-1} for shrub coppice dune, 69 Mg ha^{-1} yr^{-1} for semi-fixed dune fields, 31 Mg ha^{-1} yr^{-1} for dry farmlands, and 22 Mg ha^{-1} yr^{-1} for grasslands (Yan et al., 2000).

3.6 Management of Wind Erosion

The reduction in wind erosion rates in the USA since the "Dust Bowl" shows that refined understanding of soil and wind dynamics, prudent soil management, and use of conservation practices could reverse the severity of wind erosion. In the rest of the world, wind erosion rates have, however, increased or remained the same, which warrants increased research on site-specific conservation practices. Adoption of appropriate farming practices can offset wind erosion. Intensive tillage, summer fallow, and residue burning are practices that increase wind erosion. Vegetative barriers, strip cropping, continuous cropping, crop rotations, no-till, minimum tillage, management of crop residues, cover crops, green manures, animal manure, and forages are recommended practices. Practices that stabilize soil aggregates and roughen the soil surface also control wind erosion. Wind erosion prevails in large and flat fields with smooth, bare, loose, dry and non-aggregated soils.

Some of the strategies to control wind erosion include:

1. Maintain a vegetative cover (e.g., cover crops, residues)
2. Reduce cultivation during fallow
3. Establish windbreaks (trees and shrubs)
4. Reduce intensive grazing
5. Minimize or eliminate tillage
6. Reduce tillage speed and do not bury residues
7. Implement strip cropping and mulch tillage
8. Apply soil stabilizers or conditioners
9. Roughen the soil surface and reduce field length

3.7 Windbreaks

One of the most traditional methods of controlling wind erosion is the establishment of windbreaks (Fig. 3.7). Windbreaks are strips of trees, shrubs or tall grass species planted around agricultural fields, houses, and animal farms to reduce the wind velocity and erosion. Windbreaks are also referred to as wind barriers and shelterbelts.

Fig. 3.7 Well-designed windbreaks in North Dakota reduce wind erosion (Courtesy USDA-NRCS)

Any buffer strip (riparian buffers, filter strips, and grass barriers) particularly those under tall and robust plant species can serve as windbreaks for trapping wind-borne sediment and chemicals. The windbreaks protect crops from strong winds, divert the wind direction, and reduce wind velocity, thereby reducing soil erosion. By intercepting the erosive energy of the winds, windbreaks help to improve soil properties by reducing evaporation, promoting soil water storage, and reducing losses of nutrient-rich fine soil particles.

Establishing windbreaks leads to direct and indirect benefits (Table 3.2). Among the direct benefits are the decrease in wind velocity and turbulence and improvement in landscape beauty and land value while reducing the overall wind erosivity. Integration of windbreaks with row crops restores biodiversity and provides organic materials to soil. Indirect effects include changes in soil-water-crop microclimate conditions. A small decline in wind velocity by adoption of control practices results in a large decrease in wind erosion. The effectiveness of windbreaks for reducing wind erosion depends on the interaction of *wind* (e.g. velocity, intensity, direction, turbulence) and *windbreaks* (e.g., height, density, width, length, shape, vegetation

Table 3.2 Benefits of windbreaks

Soil Erosion	Soil Properties	Crops and Livestock
• Provide litter and residue cover • Slow wind velocity • Reduce losses of runoff, soil and nutrients • Reduce dust formation • Reduce off-site transport of pollutants	• Reduce temperature fluctuations • Increase soil water content • Improve soil aggregation • Increase biological activity • Increase soil organic matter content	• Protect crops and livestock • Improve crop yields • Improve landscape scenery • Reduce snow drifting

type) *characteristics* determines the rate and magnitude of wind erosion. Following the Dust Bowl in 1930's, windbreaks were established at large-scale in the Great Plains and neighboring regions in the USA (Bates, 1934).

Design and management of windbreaks vary depending on the plant barrier species, intensity of wind erosion, and soil conditions (Nordstrom and Hotta, 2004). Vegetation for windbreaks can be either planted or left behind after clearing lands for agriculture. Height, density, width, and orientation of the vegetation are critical factors for the design and effectiveness of windbreaks. Site preparation, weed control during establishment, and pruning are some of the practices for managing windbreaks. Multiple row-windbreaks reduce wind erosion more than single rows (Fig. 3.8). Windbreaks must be established perpendicular to the dominant wind direction to provide the highest protection. Height of barrier is the most important determinant of wind velocity reduction while width of barrier determines the size of the protected area. Most of the suspended soil particles are deposited at distances >9 m past the field edge (Hagen, 2005). Thus, windbreaks or conservation buffers (e.g., grass barriers, filter strips) below croplands must be wide enough (>9 m) to effectively trap some of the suspended dust in air.

3.7.1 Reduction in Wind Velocity

The reduction of wind velocity in the leeward side (away from the wind) depends primarily on the height of barriers (Fig. 3.8). A windbreak reduces the wind velocity for a distance of 30–35 times the windbreak height in the leeward side and about 5 times in the windward side (Nordstrom and Hotta, 2004). This means that a tree of 10 m height would reduce the wind velocity to a distance of 300–350 m in the

Fig. 3.8 Three-row windbreaks are effective soil erosion control measures (Photo by H. Blanco)

leeward side and 50 m in the windward side. Velocity reductions for windbreaks are about 70% at a distance of about 10× barrier height and 20% at a distance of about 20× barrier height, and the greatest reduction in wind velocity occurs within 4– 6× barrier height in the lee (Vigiak et al., 2003). The friction velocity reduction (f_{xh}) of wind in m s^{-1} by windbreaks is calculated as follows:

$$f_{xh} = 1 - \exp\left[-axh^2\right] + b\exp\left[-0.003\,(xh + c)^b\right] \tag{3.9}$$

$$a = 0.008 - 0.17\theta + 0.17\theta^{1.05} \tag{3.10}$$

$$b = 1.35\exp\left(-0.5\theta^{0.2}\right) \tag{3.11}$$

$$c = 10\,(1 - 0.5\theta) \tag{3.12}$$

$$d = 3 - \theta \tag{3.13}$$

$$\theta = op + 0.02\frac{w}{h} \tag{3.14}$$

where xh is the distance to the windbreak parallel to the wind direction in barrier heights, θ is the barrier porosity, op is the optical porosity, w is the barrier width, and h is the barrier height (Vigiak et al., 2003). The WEPS model uses Eq. (3.15) and (3.16) for high and medium/low windbreak porosities, respectively (Hagen, 1996)

$$f_{xh} = 1 - \exp\left[-0.006xh^2\right] + 0.913\exp\left[-0.033\,(xh + 4)^{1.52}\right] \tag{3.15}$$

$$f_{xh} = 1 - \exp\left[-0.0486xh^2\right] + 0.617\exp\left[-0.000165\,(xh + 5)^{4.66}\right] \tag{3.16}$$

The barrier density and porosity interacts with height to provide higher reductions in wind velocity. Uniformity in roughness is important to the effectiveness of wind barriers. Gaps within barriers do not only diminish the effectiveness but can actually increase wind velocity over the upwind velocity and funnel concentrated flow loaded with sediment.

Example 2. Estimate the friction velocity reduction at a distance of 6h from a windbreak with 0.4 of porosity. The barrier height is 6 m.

$$xh = 6h = 36$$

$$a = 0.008 - 0.17\theta + 0.17\theta^{1.05} = 0.008 - 0.17 \times 0.4 + 0.17 \times 0.4^{1.05} = 0.00495$$

$$b = 1.35\exp\left(-0.5\theta^{0.2}\right) = 1.35\exp\left(-0.5 \times 0.4^{0.2}\right) = 0.890$$

$$c = 10\,(1 - 0.5\theta) = 10\,(1 - 0.5 \times 0.4) = 8$$

$$d = 3 - \theta = 3 - 0.4 = 2.6$$

$$f_{xh} = 1 - \exp\left[-0.00495 \times (36)^2\right] + 0.89\exp\left[-0.003\,(36 + 8)^{0.89}\right] = 1.814\,\mathrm{ms}^{-1}$$

The influence of windbreaks on soil erodibility of the sheltered fields can be estimated by (Woodruff and Zingg, 1952)

$$d = 17h\left(\frac{V_m}{V}\right)\cos\theta \tag{3.17}$$

where d is distance of full protection by the barrier in the lee (m), h is height of the barrier (m), V_m is minimum wind velocity at 15 m height needed to move the most erodible soil fraction, V is actual wind velocity (m s^{-1}), and $\cos\theta$ is angle of the prevailing wind direction. The minimum velocity required to initiate soil movement on a bare, smooth, and uniform field is 9.61 m s^{-1} at 15 m height prior to soil wetting and crusting (Chepil, 1959). Under these conditions, Eq. (3.17) is reduced to

$$d = \frac{163.4h}{v} \tag{3.18}$$

Example 3. Determine the distance of full protection from the wind if the maximum wind velocity that could be tolerated is 20 m s^{-1} and the height of the barrier is 10 m. The angle of deviation of wind direction from the perpendicular to the windbreak is 25 degrees.

$$d = 17h\left(\frac{V_m}{V}\right)\cos\theta = 17 \times 10\left(\frac{9.6}{20}\right)\cos(25^0) = 73.95\,m$$

3.7.2 Density and Porosity

Density is the ratio of solid plant parts to the total area of the buffer. Barrier densities >80% create downwind turbulence and reduce the effectiveness of the windbreaks, whereas densities <20% do not adequately reduce the wind velocity. Thus, densities between 20 and 80% are recommended with an optimum of 50%. Porosity of buffers which is the ratio of pore space to the space occupied by plants (e.g., tree trunks, branches, grass stems, and leaves) is the most important attribute of windbreaks because it influences rates of air flow through the barriers and diverted flow by the barriers. This property is, however, difficult to accurately characterize owing to the tri-dimensionality of pore space within barriers. Optical porosity is a common approach to measure barrier porosity based on plant silhouettes. This approach involves the use of digitized photographic silhouettes (Vigiak et al., 2003).

3.7.3 Side-Benefits

Windbreaks not only reduce soil erosion but also protect livestock and enhance crop production. They also reduce snow drifting, protect farmsteads, enhance wildlife habitat, and develop microclimates. A microclimate zone develops on the area downwind as a result of alterations in wind velocity. Temperature within the protected area may be 2°C–3°C higher than in the unprotected areas. These alterations in temperature can significantly increase relative humidity, reduce evaporation, and enhance soil water storage. The microclimate zone may create favorable conditions for growing wind-sensitive crops. Well-designed windbreaks directly reduce wind erosion while creating a microenvironment favorable for the production of livestock and crops. Windbreaks established in erosion prone areas enhance programs of afforestation and restore marginal and degraded lands. Performance and benefits of windbreaks are nevertheless site-specific and depend on soil, barrier species, and climate conditions.

3.7.4 Constraints

While benefits of windbreaks barriers to wind erosion control are numerous, large-scale establishment of barriers in wind erosion-affected regions, particularly in the developing world, is limited. Reasons for the slow adoption include the following (Sterk, 2003):

- lack of tree/shrub material
- lack of management guidelines
- absence of technical support, training, and external financial
- lack of incentives from conservation programs
- concerns over reduction of cropping area
- lack of land ownership
- high costs of establishment
- invasion of insects and pests
- attraction of birds which consume grains and reduce yields
- competition of barriers with crops for light, water, and nutrients
- difficulties in protection against grazing by livestock

3.8 Crop Residues

Crop residues left on the soil surface protect the soil against wind erosion by increasing the surface roughness. Maintaining mixtures of standing stalks and broken coarse residues on the soil surface is the most effective practice to control wind erosion. Wind erosion decreases in direct proportion with the amount of residues present on the soil surface. Residue mulch buffers wind energy and serves as a natural blanket between the erodible soil surface and the wind. Residues reduce wind erosion by altering the wind velocity profile near the soil surface, by absorbing the wind energy, and thereby increasing the threshold wind velocity required to cause soil erosion. Residues not only change the aerodynamic forces of wind but also improve the soil structural properties. By reducing excessive evaporation and trapping rain and snow, residues increase the soil water content.

Crop residues provide an emergency control when applied in freshly disturbed soils prone to wind erosion. Severe wind erosion occurs only when the soil surface is unprotected. Residues or bales of straw must be applied and spread all over the upwind edge of erodible areas. Intensively tilled crests and shoulders facing wind at right angle require emergency control to reduce the erosion hazard.

3.8.1 Flat and Standing Residues

Standing stalks are at least five times more effective for controlling wind erosion than flat residues (Fryrear and Bilbro, 1998). Vertical stems not only intercept saltating particles and suspended particles floating near the soil surface but also anchor the soil. Flat residues are readily blown away by wind. The effectiveness of standing stalks depends on the number of stems, density of stems, diameter of stems, leaf fractions, and stem area index. Flat residues in contact with the soil are also decomposed more rapidly than a comparable amount of standing residues (Lopez et al., 2003). The first order decay model is a common approach to determine the decomposition rates of standing and flat residues (Steiner et al., 1994). The slowly decomposing residue mulch cover enhances formation of stable soil aggregates and accumulation of soil organic matter. Traffic and tillage practices that flatten crop residues must be reduced.

3.8.2 Availability of Residues

While the effectiveness of residue mulch in controlling erosion is widely recognized, a major constraint, however, is the limited availability of crop residues in arid and semiarid regions. The limited amount of residues is used for livestock forage and other purposes. The removal of residue after harvest increases wind erosion in dry regions. Leaving residues as much as possible and reducing intensive grazing of residues after harvest are the best options for conserving water and thus reducing wind erosion. Burning of residues must be eliminated to reduce destruction of soil humus and reduction in organic matter concentration. Practices that increase production of residues and enhance residue accumulation on the soil surface are the most cost-effective measures of reducing wind erosion.

3.9 Perennial Grasses

Traditional soil conservation techniques against wind erosion include mulching with crop residues or tree branches and application of animal manure. New approaches include establishing tree and grass cover and enhancing regeneration of natural woody vegetation. Growing perennial native grasses is an effective strategy for wind erosion control because these plant species develop extensive and deep root systems which stabilize and anchor loose and erodible soil while producing large amounts of aboveground biomass. Orientation, density, width, species, and age of grasses influence their effectiveness for erosion control. Close- and dense-growing grass species protect the soil better than sparse tall bushes or shrubs. Grasses tend to accumulate and absorb blowing soil particles and reduce off-site transport of windblown materials. Converting degraded lands to pasture or meadow is a useful strategy to restore soil fertility and reduce wind erosion (Sterk, 2003). Because complete removal of crops at harvest leaves the soil bare, growing perennial grasses provides a permanent cover for erosion control. If a land must be cultivated to row crops, perennial forage or native vegetation can be used in rotation with row crops.

The grasslands must be managed under controlled stocking rates to reduce overgrazing. Intensive grazing is the main cause of wind erosion in grasslands. On sandy soils prone to wind erosion in northern China, continuous grazing decreased ground cover, reduced soil organic matter content, and increased sand content due to wind erosion (Su et al., 2005). Exclusion or reduction in grazing intensity enhances grass recovery, biomass accumulation, and growth of annual and perennial grasses. It is particularly critical during early stages of grass establishment. Growing perennial grasses rather than annual grasses is more effective at restoring degraded soils. In the USA, switchgrass, big bluestem, and Indian grass are some of the native tall grass prairies that can be used o reduce wind erosion. These deep-rooted grasses stabilize movement of sand dunes.

3.10 Conservation Tillage

Conservation tillage practices are important options to conserve soil water and produce abundant residues. Continuous cropping with annual and perennial plant species must be practiced on all cultivated soils to reduce risks of wind erosion. Type of tillage directly influences soil roughness and amount of crop residues left on the soil surface. Timing of tillage and type of tillage implements determine the distribution and burial of crop residues. Tillage must be designed in a way that large amounts of residue are left on the soil surface. Improved cropping systems combined with reduced tillage can reduce airborne pollution by as much as 50–95% as compared to plow till age (Upadhyay et al., 2002). Tillage dries out the exposed soil clods and increases their susceptibility to erosion. If a field must be tilled, avoid tilling knolls and maintain tractor speed at $<8\,\mathrm{km\,h^{-1}}$. Slow tillage minimizes soil pulverization and promotes formation of large clods. Large and wide blades reduce residue burial. One pass with moldboard plow leaves $<10\%$ of residue cover while the same pass with chisel plow leaves about 60%, and with wide blades about 90%.

No-till. No-till management is a conservation-effective strategy to reduce wind erosion because it leaves most of the residues and maintains an undisturbed soil surface. It improves soil water storage, reduces evaporation, and decreases desiccation. Moist soils are less susceptible to erosion. No-till is not, however, always the best choice for clayey and poorly drained soils, which are susceptible to compaction and hardsetting. These soil may require some additional tillage (e.g., subsoiling) to reduce no-till management-induced soil compaction. A combination of reduced tillage and crop residue management is an option in arid and semiarid regions to reduce wind erosion while maintaining crop production. Conversion of plow till to no-till can reduce wind erosion by about 80% (Wang et al., 2006). No-till not only reduces wind erosion but increases soil organic matter and nutrient content. In the Great Plains of the USA, intensive no-till cropping systems combined with high-residue producing crops are adequate practices to control wind erosion (Cantero-Martinez et al., 2006). Because traditional cropping systems such as winter wheat-summer fallow systems in the region do not leave sufficient residue to reduce high evapotranspiration and wind erosion, there is a shift from plow till with wheat and fallow to intensive no-till to reduce wind erosion.

Stubble-mulch tillage. Stubble-mulch tillage is a conservation tillage that loosens the soil, minimizes soil inversion, and leaves crop residues on the surface. It is a form of subsurface tillage without burying crop residues. In semiarid soils under winter wheat in the Pacific Northwest, soil water storage was found to be the highest under stubble-mulch tillage, followed by the bare soil and no-till systems. High residue cover in the no-till systems reduces evaporation and increases water storage in spring, but water losses from no-till fallow can be higher than those from stubble-mulch tillage (Schillinger and Bolton, 1993). Stubble-mulch tillage may be more effective at increasing water storage and improving soil aggregation than no-till systems with complex rotations (e.g., wheat-grain sorghum-fallow) in regions where production of residues is low.

Cover crops. Cover crops provide multiple benefits to disturbed soils. They protect soil from wind erosion and also improve soil properties by stabilizing aggregates and promoting biological activity. Cover crops are often used to protect soil between harvest and planting season when soils remain bare. Planting cover crops between rows of main crops protects sensitive plants from wind damage during the growing season. Cover crops can, however, compete with main crops for limited water in arid and semiarid regions and are thus most suitable for temperate and humid climates with high precipitation. Cover crops provide additional residue cover in regions with limited return of crop residues. Cover crops of small grain planted in the spring or the fall are suitable practices.

Strip cropping. Strip cropping consists of establishing alternate strips of vegetation with high and low wind protective ability. Strips of row crops can be alternated with annual or perennial grass strips to protect erodible landscapes. Recommended strip widths can be 24 m in loamy sands, 50 m in silt loams, and 96 m in silty clay loams. Strips must be established perpendicular to the wind direction. The width of strips depends on the sensitivity of crops to sand blasting. Narrower strips protect sensitive crops better. Strip widths must also be designed to accommodate the farm equipment for maneuvering and turns.

Other measures. Crop rotations, green manuring, fall seeded cover crops, field borders, and contour farming are additional practices to reduce wind erosion. Rapid and effective measures include crop residue spreading and mulch tillage. Application of organic amendments such as animal manure and compost is also an option. Organic amendments halt wind erosion by improving binding and stabilizing soil aggregates, improving soil fertility, and increasing soil water retention. Emergency measures to control erosion include building up of the cloddiness of soil surface by ripping and covering immediately the disturbed soil with crop residues and manure.

Summary

Rates of wind erosion can be as high as those from water erosion. Anthropogenic and climatic processes influence the magnitude of wind erosion. Deforestation and intensive plowing are the main causes of accelerated wind erosion. The processes of wind erosion are similar to those of water erosion: detachment, transport, and deposition. Transport of soil particles by wind is a function of particle size. Soil particles are transported by wind through *suspension, saltation*, and *surface creep*. Smaller soil particles remain suspended in air longer time and are transported faster and farther as atmospheric dust. About 50 to 70% of soil particles are transported by saltation, 30 to 40% by suspension, and about 5 to 25% by surface creep. Climatic conditions (e.g., wind speed and duration, precipitation, air humidity, temperature), land surface properties (field slope, length, width, roughness, residue management,), soil characteristics (e.g., particle size distribution, particle density, aggregate size and stability, water content, organic matter content), and land use and management (e.g., forest, rangeland, pasture, type of cultivation, windbreaks) are factors that determine the severity of wind erosion.

Wind erosion rates are measured using *active* and *passive* samplers. Passive samplers are less expensive and more portable than active samplers. The Bagnold trap, the Modified Wilson and Cooke trap, and the Big Spring Number Eight (BSNE) are the most popular samplers for collecting dust and saltating particles. The smaller the particles, the greater the difficulty of collecting representative samples. The wind-tunnel is a common method used to directly measure wind erosion under lab and field conditions. Similar to tillage erosion, radionuclide fallouts (e.g., ^{137}Cs) are also used for quantifying wind erosion rates and study the spatial distribution of wind erosion patterns across large areas. Wind erosion is manageable with appropriate farming practices. While intensive tillage, summer fallow, and residue removal increase wind erosion, best management practices such as vegetative barriers, strip cropping, crop rotations, no-till, reduced tillage, residue mulch, cover crops, and growing perennial crops reduce wind erosion. Windbreaks are the most traditional means to reduce wind velocity. Height and width of windbreaks and density of vegetation influence the effectiveness of windbreaks.

Study Questions

1. Estimate the wind velocity at 10, 15, and 30 m above the soil surface if the wind velocity at the 2.5 m height is 3 m s^{-1} and height of crop is 1 m. Assume z_0 is equal to 0.13 the height of crops.

2. Compute the friction velocity of wind flowing thorough a windbreak barrier of 0.3 of porosity at a distance of 5 h (5 the barrier height) from the windbreak given that the barrier height is 4 m.
3. Estimate the surface crust factor for a clay loam with 6% of sand, 40% of silt, and 2.8% of organic C.
4. Estimate the amount of soil lost from a 600 m long field under winter wheat which has 20% of non-erodible knolls, 80 mm tall ridges with a spacing of 600 mm, 3 Mg ha^{-1} of residue standing on the soil surface. The factor C value is 75%.
5. Determine the maximum wind velocity at 15 m height for a distance of full protection of 85 m for a barrier height of 11 m. The angle of deviation of wind direction from the perpendicular to the windbreak is 30 degrees.
6. Calculate the spacing between windbreaks of height 12 m for a bare, smooth, and uniform field if the wind velocity at 15 m is 14 m s^{-1}. Assume that the direction of prevailing wind is perpendicular to the windbreak.
7. Calculate the distance of full protection from the wind by strips of corn on a strip cropping system if the wind velocity is 8.5 m s^{-1}. The height of corn plants is 1.8 m. The angle of deviation of wind direction from the perpendicular to the windbreak is 20 degrees.
8. Discuss the strategies of wind erosion management.
9. Describe the benefits of windbreaks to crop and livestock production as well as environmental quality.
10. Discuss the leeward and windward sides of the windbreaks, and draw a windbreak to explain your answer.

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Chapter 4
Modeling Water and Wind Erosion

4.1 Modeling Erosion

Modeling water and wind erosion is important to understanding the processes governing soil erosion, predicting runoff and soil erosion rates, and identifying or choosing appropriate measures of erosion control. Modeling permits the: (1) understanding of the driving processes, (2) evaluation of on-site and off-site impacts on soil productivity and water and air pollution on large scale, (3) identification of strategies for erosion control, and (4) assessment of the performance of soil conservation practices for reducing water and wind erosion. Well-developed and properly calibrated models provide good estimates of soil erosion risks. Soil erosion results from a complex interaction of soil-plant-atmospheric forces. Thus, modeling soil erosion requires a multidisciplinary approach among soil scientists, crop scientists, hydrologists, sedimentologists, meteorologists, and others. Models must be able to integrate processes, factors and causes at various spatial and temporal scales.

Numerous models of differing prediction capabilities and utilities have been developed. The advent of technological tools such as remote sensing and GIS has significantly enhanced the usefulness of soil erosion models. The coupling of GIS and remote sensing with empirical and process-based soil erosion models has improved their predictive capability. The GIS stores the essential database needed as input for modeling erosion and elaboration of maps of erosion-affected areas. Remote sensing is, for example, useful to estimate land cover over large geographic areas, which is a critical input for modeling erosion. Remote sensing and GIS tools also allow the scaling up of modeled data from small plots (e.g., USLE) to large areas. Modeling soil erosion involves integration of complex and variable hydrological processes across large areas to understand the magnitude of soil erosion. There are empirical and process-based models to estimate soil erosion at various scales (e.g., plot, watershed, field).

4.2 Empirical Models

The USLE technology comprises a set of empirical equations to predict soil loss as follows:

4.3 Universal Soil Loss Equation (USLE)

The USLE developed in the USA is the most widely used empirical model worldwide for estimating soil loss (Wischmeier and Smith, 1965). Information from the USLE is used in planning and designing conservation practices. This model is not strictly based on hydraulic principles and soil erosion theory. It thus simplifies the processes of soil erosion. The USLE was specifically intended to predict soil loss from cultivated soils under specific characteristics. It has sometimes been used inappropriately and applied to soil and land use conditions different from those for which it was developed. It provides a long-term annual average estimate of soil loss from small plots or field segments with defined dimensions. The USLE was developed from measured data rather from physically-based modeling approaches. The limited consideration of all the complex and interactive factors and processes of soil erosion with the USLE limits its applicability of USLE to all conditions.

The USLE is, however, advantageous over sophisticated models because it is simple, easy to use, and does not require numerous input parameters or extensive data sets for prediction. The simplicity of the equation for its practical use has sacrificed accounting for all the details of soil erosion. Parameters are estimated from simple graphs and equations. Unlike process-based models, the USLE can not simulate the following:

- Runoff, nutrient, and soil loss from watersheds or field-scale areas.
- Soil loss on an event or daily basis and variability of soil loss from storm to storm.
- Interrill, rill, gully, and streambank erosion separately.
- Processes of concentrated flow of flow channelization and sediment deposition.
- Detailed processes (e.g., detachment, transport, and deposition).

The average annual soil loss is estimated as

$$A = R \times K \times LS \times C \times P \tag{4.1}$$

where A is average annual soil loss (Mg ha^{-1}), R is rainfall and runoff erosivity index for the location of interest, K is erodibility factor, LS is topographic factor, C is cover and management factor, and P is support practice factor. The early versions of USLE were exclusively solved using tables and figures (e.g., nomographs). The continued improvement has resulted in MUSLE and Revised USLE (RUSLE 1 and 2).

4.3.1 Rainfall and Runoff Erosivity Index (EI)

The EI is computed as the product of total storm energy (E) times the maximum 30-min intensity (I_{30}) of the rain.

$$EI = E \times I_{30} \tag{4.2}$$

The USLE uses the annual *EI* which is computed by adding the *EI* values from individual storms that occurred during the year. According to Wischmeier and Smith (1978), the EI corresponds closely with the amount of soil loss from a field. The EI as used in the USLE overestimates the *EI* for tropical regions with intensive rains. The USLE-computed EI is only valid for rain intensities $\leq$63.5 mm h^{-1}. Modifications to EI have been proposed for tropical regions (Lal, 1976). The 30-minute intensity for a given storm and location is obtained from rain gauge charts recording the 30-minute with the largest amount of rainfall. Data on R for different locations of the continental USA and estimates for the world are available (Foster et al., 1981). In the USA, about 4000 sites were analyzed for their rainfall intensities for a range of rain-return periods to develop an iso-erodent map (Fig. 4.1). Values of EI_{30} below 50 correspond to dry regions (e.g., Great Plains) and those above 500 correspond to humid regions.

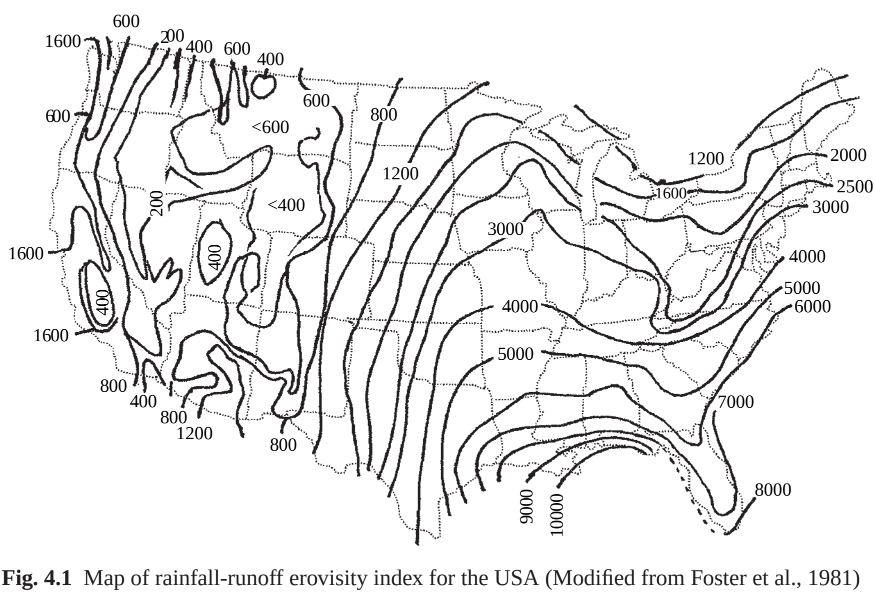

Fig. 4.1 Map of rainfall-runoff erovisity index for the USA (Modified from Foster et al., 1981)

4.3.2 Soil Erodibility Factor (K)

Soil erodibility refers to soil's susceptibility to erosion. It is affected by the inherent soil properties. The K values for the development of USLE were obtained by direct measurements of soil erosion from fallow and row-crop plots across a number of sites in the USA primarily under simulated rainfall. The K values are now typically obtained from a nomograph (Foster et al., 1981) or the following equation:

$$K = \frac{0.00021 \times M^{1.14} \times (12 - a) + 3.25 \times (b - 2) + 3.3 \times 10^{-3}(c - 3)}{100} \quad (4.3)$$

$$M = (\%\, silt + \%\, very\ \ fine\, sand) \times (100 - \%\ \text{clay}) \quad (4.4)$$

where M is particle-size parameter, a is % of soil organic matter content, b is soil structure code (1 = very fine granular; 2 = fine granular; 3 = medium or coarse granular; 4 = blocky, platy, or massive), and c profile permeability (saturated hydraulic conductivity) class [1 = rapid (150 mm h^{-1}); 2 = moderate to rapid (50–150 mm h^{-1}); 3 = moderate (12–50 mm h^{-1}); 4 = slow to moderate (5–15 mm h^{-1}); 5 = slow (1–5 mm h^{-1}); 6 = very slow (<1 mm h^{-1})]. The size of soil particles for very fine sand fraction ranges between 0.05 and 0.10 mm, for silt content between 0.002 and 0.05, and clay <0.002 mm. The soil organic matter content is computed as the product of percent organic C and 1.72.

4.3.3 Topographic Factor (LS)

The USLE computes the LS factor as a ratio of soil loss from a soil of interest to that from a standard USLE plot of 22.1 m in length with 9% slope as follows:

$$LS = \left(\frac{Length}{22.1}\right)^m \left(65.41 \sin^2\theta + 4.56 \sin\theta + 0.065\right) \quad (4.5)$$

$$m = 0.6\left[1 - \exp\left(-35.835 \times S\right)\right] \quad (4.6)$$

$$\theta = \tan^{-1}\left(\frac{S}{100}\right) \quad (4.7)$$

where S is field slope (%) and θ is field slope steepness in degrees.

4.3.4 Cover-Management Factor (C)

The C-factor is based on the concept that soil loss changes in response to the vegetative crop cover during the five crop stage periods: *rough fallow, seedling, establishment, growing*, and *maturing crop*, and *residue* or *stubble*. It is computed as the soil loss ratio from a field under a given crop stage period compared to the loss from a field under continuous and bare fallow conditions with up- and down-slope tillage (Wischmeier and Smith, 1978). Crop type and tillage method, the two sub-factors defining the C, are multiplied to compute the C-values. Estimates of C values for selected vegetation types are shown in Table 4.1. Detailed calculations of C values are presented by Wischmeier and Smith (1978).

Table 4.1 C values for some tillage and cropping systems (After Wischmeier and Smith, 1978)

Vegetation	Description	C values
Grain corn	Moldboard plow, no residues, plowed during:	
	– fall	0.40
	– spring	0.36
	Mulch tillage	0.24
	Chisel plow, >50% residue cover, spring plowing	0.20
	Ridge tillage	0.14
	No-till with 100% residue cover	0.05
Corn silage and beans	Moldboard plow, no residues, plowed during:	
	– fall	0.50
	– spring	0.45
	Mulch tillage	0.30
	Ridge tillage	0.17
	No-till with 100% residue cover	0.10
Cereals	Fall plowed	0.35
	Spring plowed	0.32
	Mulch tillage	0.21
	Ridge tillage	0.12
	No-till with 100% residue cover	0.08
Corn-soybean rotation	Moldboard plow, no residues, fall plowing	0.50
	Chisel plow, >50% residue cover, spring plowing	0.23
	No-till with 100% residue cover	0.05
Corn-soybean rotation	Moldboard plow, no residues, fall plowing	0.20
	Chisel plow, >50% residue cover, spring plowing	0.14
	No-till with 100% residue cover	0.05
Hay and pasture	Dense stand of sod-like grass	0.02
Forest	>90% canopy cover and 100% litter cover	0.001
Short and managed trees without understory vegetation (fruit trees)	At least 75% of canopy cover without litter cover	0.35
	At least 75% of canopy cover with about 30% litter cover	0.08

4.3.5 Support Practice Factor (P)

The P-factor refers to the practices that are used to control erosion. It is defined as the ratio of soil lost from a field with support practices to that lost from a field under

Table 4.2 P values for contouring and strip-cropping (After Wischmeier and Smith, 1978)

Land slope (%)	Contouring		Strip cropping				
	P value	Maximum slope length (m)	P value			Strip width (m)	Maximum slope length (m)
			A	B	C		
1–2	0.60	122	0.3	0.45	0.60	40	243
3–5	0.50	91	0.25	0.38	0.50	30	182
6–8	0.50	61	0.25	0.38	0.50	30	122
9–12	0.60	36	0.30	0.45	0.60	24	74
13–16	0.70	24	0.35	0.52	0.70	24	49
17–20	0.80	18	0.40	0.60	0.80	18	36
20–25	0.9	15	0.45	0.68	0.90	15	30

Table 4.3 P values for combined support practices [After Wischmeier and Smith (1978) and USDA-ARS (1997)]

Land slope (%)	Contour factor	Strip crop factor	Terrace factor		
			Terrace interval (m)	Closed outlets	Open outlets
1–2	0.60	0.30	33	0.5	0.7
3–8	0.50	0.25	33–44	0.6	0.8
9–12	0.60	0.30	43–54	0.7	0.8
13–16	0.70	0.35	55–68	0.8	0.9
17–20	0.80	0.40	69–60	0.9	0.9
21–25	0.90	0.45	90	1.0	1.0

up-and down-slope tillage without these practices. The P values vary from 0 to 1 where the highest values correspond to a bare without any support practices.

Maintaining living and dead vegetative cover and practicing conservation tillage significantly reduces soil erosion. The combined use of various practices is more effective than a single practice for controlling erosion in highly erodible soils. In such a case, support practices (P) including contouring, contour stripcropping, terracing, and grass waterways must be used. The P values are obtained from Tables 4.2 and 4.3.

In systems with various support practices, P values are calculated as follows:

$$P = P_c \times P_s \times P_t \tag{4.8}$$

where P_c is contouring factor for a given field slope, P_s is strip cropping factor, and P_t is terrace sedimentation factor (Table 4.3).

Example 1. A 130 m long field with 5% slope is under continuous corn managed with chisel plowing in eastern Ohio. The soil is silt loam (10% coarse and medium sand, 10% very fine sand, 20% clay, and 60% silt) with 2.5% of soil organic matter content. The structure is fine granular and the saturated hydraulic conductivity is 40 mm h^{-1}. Estimate the average annual soil loss if the field is contoured and strip cropped with no terraces.

1. **Rainfall Erosivity**
   From Fig. 4.1, R = 2100.

2. Soil Erodibility

$$K = \frac{2.8 \times 10^{-5} \times M^{1.14} \times (12 - a) + 0.43 \times (b - 2) + 0.33 \times (c - 3)}{100}$$
$$M = (\% silt + \% very\ fine\, sand) \times (100 - \% clay) = (80 + 10) \times (100 - 20) = 7200$$
$$a = 2.5$$
$$b = 3$$
$$c = 3$$
$$K = \frac{2.8 \times 10^{-5} \times (7200)^{1.14} \times (12 - 2.5) + 0.43 \times (3 - 2) + 0.33 \times (3 - 3)}{100} = 0.0707$$

3. Topographic Factor

$$m = 0.6\left[1 - \exp(-35.835 \times S)\right] = 0.6[1 - \exp(-35.835 \times 0.05) = 0.4999$$
$$\theta = \tan^{-1}\left(\frac{S}{100}\right) = \tan^{-1}(0.05) = 2.862$$
$$LS = \left(\frac{130}{22.1}\right)^m \left(65.41 \sin^2 2.862 + 4.56 \sin(2.862) + 0.065\right)$$
$$= 2.425 \times 0.456 = 1.106$$

4. Cover-Management Factor
C value for continuous corn under chisel plow = 0.20

5. Support Practice Factor:

$$P_c = 0.50$$
$$P_s = 0.25$$
$$P_t = 1 \text{ for no terraces}$$
$$P = P_c \times P_s \times P_t = 0.50 \times 0.25 \times 1 = 0.125$$
$$A = R \times K \times LS \times C \times P = 2100 \times 0.0707 \times 1.106 \times 0.20 \times 0.125 = 4.1\,\text{Mg}\,\text{ha}^{-1}$$

Example 2. Estimate the soil loss if the cropped field had been managed without contouring and strip cropping?

Under these new conditions, P value would be equal to 1.

Thus,
$$A = R \times K \times LS \times C \times P = 2100 \times 0.0707 \times 1.106 \times 0.20 \times 1 = 32.84\,\text{Mg}\,\text{ha}^{-1}$$

The elimination of contouring and strip cropping dramatically increased the average annual soil loss by about 8 times.

4.4 Modified USLE (MUSLE)

The MUSLE is a modified form of USLE. While USLE predicts sediment yield based on rainfall, MUSLE predicts it by using runoff factor, which accounts for the antecedent soil water content. This modification allows the use of USLE for predicting sediment loss on a storm event basis.

$$Sed = 11.8 \left(Q \times q_p \times A\right)^{0.56} \times K \times C \times P \times LS \times CFRG \qquad (4.9)$$

where *sed* is sediment yield on a storm event basis (Mg), Q is surface runoff volume (mm), q_p is peak runoff (m^3 s^{-1}), A is area of the hydrologic response unit (HRU) (ha), and *CFRG* is coarse fragment factor, which is estimated as

$$CFRG = \exp(-0.053 \times Rock) \qquad (4.10)$$

where rock is % rock in the uppermost soil layer.

4.5 Revised USLE (RUSLE)

This model is more comprehensive and detailed than USLE and is based on empirical- and process-based approaches (Renard et al., 1997). As compared to USLE, it includes more EI values for the western U.S. in addition to those in the eastern U.S. It incorporates soil processes (e.g., freezing and thawing) and changes in water content into the USLE. It uses computer tools to calculate complex LS interactions based on rill and interrill erosion relationships and incorporates information on canopy and surface residue cover and the effects of temperature and soil water on above- and below-ground residue decomposition at short time (1/2 month) intervals. In USLE, the C values are calculated from tables with data from field experiments, but RUSLE computes these values from four sub-factors, which are the following:

- prior land use (PLU) factor which accounts for the amount and biomass and tillage practices from previous years,
- the canopy (CC) factor accounting for the vegetative cover,
- the surface cover (SC) factor that reflects the amount of residue mulch left on the soil surface, and
- surface roughness (SR) factor.

The RUSLE accounts for the influence of farming across slopes as well as stripcropping and buffer strips within the P factor. The P values are estimated based on slope length and steepness, ridge height, soil deposition, soil infiltration, and the cover and roughness conditions. Friendly-user online assessments of soil loss for RUSLE are available to estimate soil loss by simply entering the county name, slope, length, and soil series name, and crop rotation of the cropped field. RUSLE1 and RUSLE2 compute transport capacity (T_c) as

$$T_c = k_t q_p \sin(\theta) \tag{4.11}$$

where k_t is transport capacity that depends on the hydraulic resistance of soil surface roughness and vegetative cover, and q_p is runoff rate, and θ is angle of the slope. Sediment deposition (D) is estimated as

$$D = \left(\frac{V_f}{q_p}\right)(T_c - g) \tag{4.12}$$

where is V_f is fall velocity of the sediment and g is sediment load.

4.6 Process-Based Models

The fundamental principle for all complex models and simplified equations of sediment transport prediction is the continuity equation of mass (Foster, 1982), which is

$$\frac{\partial q_s}{\partial x} + \rho_s \frac{\partial (cy)}{\partial t} = D_r + D_i \tag{4.13}$$

where q_s is sediment load, x is distance downslope, ρ_s is mass density of sediment particles, c is sediment concentration, y is flow depth, t is time, D_r is deposition rate, and D_i is sediment delivered to the rill from the interrill areas. The parameters q_s, D_r, and D_i are determined per unit width of the field. The first term, $\partial q_s/\partial x$, represents the change of sediment flow rate with respect to distance x, whereas the $\rho_s \partial(cy)/\partial t$ represents the change in sediment storage with respect to time. If the sediment flow is shallow, the storage term ($\rho_s \partial(cy)/\partial t$) may be negligible, and we have

$$\frac{\partial q_s}{\partial x} = D_r + D_i \tag{4.14}$$

4.7 Water Erosion Prediction Project (WEPP)

The WEPP is a process- and computer-based model and is part of a new generation of prediction technology (Flanagan and Nearing, 1995). It is used for hillslopes and watersheds based on fundamental principles of overland flow dynamics, infiltration, evaporation, evapotranspiration, erosion mechanics, percolation, drainage, surface ponding, interception of rainfall and runoff by plant, residue decomposition, soil consolidation, and tillage and soil management. It uses climate data from a robust file to account for mean daily precipitation, maximum and minimum temperature, mean daily solar radiation, and mean direction and speed of wind, and other climate factors. WEPP can predict soil erosion on a storm event and continuous basis for diverse tillage and cropping systems (e.g., crop rotations, terracing, contouring, strip cropping).

The advantage of WEPP over other erosion models is that it can estimate erosion for single hillslopes (hydrologic units) and the whole watershed which comprises various hillslopes. It simulates soil erosion at different temporal (daily, monthly, annual basis) and spatial (hillslope, small, medium, and large watersheds) scales. It simulates rill and interrill erosion over hillslopes and sediment transport and deposition in channels and impoundments interaction with surface cover conditions, soil properties, surface roughness, and soil management.

The main components of the model are (Flanagan and Nearing, 1995):

1. weather conditions
2. winter processes
3. irrigation practices
4. infiltration dynamics
5. overland flow hydraulics
6. water balance
7. plant growth
8. plant residue decomposition
9. soil parameters
10. hillslope erosion and deposition
11. watershed channel hydrology
12. watershed impoundment component

A brief overview of selected equations used in WEPP is presented below. A complete description of the WEPP model components is presented by Flanagan and Nearing (1995).

The peak intensity of a storm is computed as follows (Nicks et al., 1995)

$$r_p = -2P \ln(1 - rl) \tag{4.15}$$

where r_p is peak intensity of the precipitation (mm h^{-1}), P is precipitation amount (mm), and rl is gamma distribution of the monthly mean half-hour precipitation amounts. The surface runoff is estimated using the kinematic wave model (Stone, 1995), which is based on the continuity equation:

$$\frac{\partial h}{\partial t} + \frac{\partial q}{\partial x} = v \tag{4.16}$$

and the depth of peak discharge is:

$$q = \alpha h^m \tag{4.17}$$

where h is runoff flow depth (m), q is runoff discharge per unit width (m^3 m^{-1} s^{-1}), α is coefficient of depth of runoff discharge, m is depth-discharge exponent, and x is distance downslope (m).

Runoff depth depends on the infiltration rate. WEPP computes infiltration based on Green-Ampt model, which uses effective hydraulic conductivity (K_e) and wetting front matric potential as input parameters (Alberts et al., 1995). When measured K_{eff} is not available, WEPP computes the 'baseline" effective hydraulic conductivity (K_b) internally based on:

$$K_b = -0.265 + 0.0086(100sand)^{1.8} + 11.46CEC^{-0.75} \tag{4.18}$$

if the soil clay content is $\leq$40%
and

$$K_b = 0.0066e^{\left(\frac{2.44}{clay}\right)} \tag{4.19}$$

if the soil clay content is $> 40\%$

The WEPP allows for corrections for the effects of temporal variables (e.g., crusting, tillage operations) on K_b.

The WEPP predicts soil erosion based on separate processes of interrill and rill erosion. The movement of sediment in WEPP hillslope model is described by the equation of sediment continuity (Foster et al., 1995) as follows

$$\frac{\partial G}{\partial x} = D_f + D_i \tag{4.20}$$

where G is sediment load ($kgs^{-1}m^{-1}$), x is distance downslope of a field (m), G is sediment load ($kg\,s^{-1}$ m^{-1}), D_f is rill erosion rate ($kg\,s^{-1}$ m^{-2}), and D_i is interrill erosion rate (kg s^{-1} m^{-2}). While D_i is always positive, the D_f is has a positive value for detachment and a negative value for deposition. The rill detachment is computed as per Eq. (4.21)

$$D_f = D_c\left(1 - \frac{G}{T_c}\right) \tag{4.21}$$

where D_c is detachment capacity by rill runoff ($kg\,s^{-1}$ m^{-2}), and T_c is sediment transport capacity ($kg\,s^{-1}$ m^{-1}). If the hydraulic shear stress of the rill is higher than the critical shear stress of the soil, D_c is described as per Eq. (4.22)

$$D_c = K_r\left(\tau_f - \tau_c\right) \tag{4.22}$$

where K_r ($s\,m^{-1}$) is a rill erodibility parameter, is τ_f hydraulic flow shear stress, and τ_c is rill detachment threshold parameter. Rill detachment does not occur when flow shear stress is lower than the critical shear stress of the soil. The net deposition in a rill is computed as per Eq. (4.23)

$$D_f = \frac{\beta V_f}{q}(T_c - G) \tag{4.23}$$

where β is a raindrop-induced turbulence coefficient assumed to be 0.5 for rain and 1.0 for snow melting and furrow irrigation, V_f is effective fall velocity of sediment particles (m s^{-1}), and q is flow discharge (m^2 s^{-1}). The sediment transport capacity (T_c) is estimated as

$$T_c = k_t \tau_f^{\frac{3}{2}} \tag{4.24}$$

where k_t is a transport capacity coefficient (m$^{0.5}$ s^2 kg$^{-0.5}$).

The WEPP model is under continuous improvement and integration with other technological advances. Now, WEPP is being linked to GIS through the Geo-spatial interface for WEPP (GeoWEPP), which allows the simulations based on digital sources (e.g, internet sources) of readily available geo-spatial information such as digital elevation models (DEM), climate data, soil surveys (e.g., USDA-NRCS data), precision farming, and topographical maps using the Arcview software (Renschler, 2003). The GIS component allows the selection, manipulation, and parameterization of potential input parameters for the simulations at small- and large-scale land areas of interest. The expansion of traditional WEPP and its combination with GIS add flexibility of WEPP. The GeoWEPP is a variant of the traditional WEPP and its further development would permit the simulation of distribution, extent, and magnitude of soil erosion at larger spatial scales and represent an improved approach for land use planning and soil and water conservation.

4.8 Ephemeral Gully Erosion Model (EGEM)

The EGEM was specifically developed to predict gully formation and erosion based on physical principles of gully bed and side-wall dynamics (Woodward, 1999; Foster and Lane, 1983). Common erosion models such as USLE, RUSLE, and WEPP do not include direct options for predicting gully erosion. The EGEM considers the dynamic processes of concentrated flow responsible for gully incision and headcut development. The EGEM is one of the few process-based models to predict gully erosion. The Chemicals, Runoff and Erosion from Agricultural Management Systems (CREAMS) is another model that can predict gully erosion by accounting for the shear of flowing water, runoff and sediment transport capacity, and changes in channel bed and side dimensions. The EGEM is a development of the Ephemeral Gully Erosion Estimator (EGEE) (Laflen et al., 1986). The EGEM consists of two major components: hydrology and erosion. The hydrologic component is estimated using the runoff curve number, drainage area, watershed slope and flow depth, peak runoff discharge, and runoff volume. The erosion component is based on the width and depth of ephemeral channels. The EGEM can predict gully erosion for single storms or seasons or cropstage periods. It assumes that soil erodes to a depth of about 45 cm (e.g., tillage, resistant layer). The width of the gullies is computed using regression equations (Foster, 1982; Woodward, 1999) as

$$W_e = 2.66\left(Q^{0.396}\right)\left(n^{0.387}\right)S^{-0.16}CS^{-0.24} \tag{4.25}$$

$$W_u = 179\left(Q^{0.552}\right)\left(n^{0.556}\right)S^{0.199}CS^{-0.476} \tag{4.26}$$

where W_e is equilibrium channel width (m), W_u is ultimate channel width (m), Q is peak runoff rate ($m^3\ s^{-1}$), n is Manning's roughness coefficient, S is concentrated runoff slope, and CS is critical shear stress ($N\,m^{-2}$). The detachment rate in gullies is computed similar to that in CREAMS by a modified form of rill erosion equation as follows

$$D = KC\,(1.35t - t_c) \tag{4.27}$$

where D is detachment rate ($g\,m^{-2}\ s^{-1}$), KC is channel erodibility factor ($g\,s^{-1}$ N^{-1}), t is average shear stress of flowing water ($N\,m^{-2}$), and t_c is critical shear stress of soil ($N\,m^{-2}$)

4.9 Other Water Erosion Models

Other models for predicting soil erosion include the Agricultural Non-Point Source pollution model (AGNPS), Annualized Agricultural Non-Point Source Pollutant Loading (AnnANPSPL), Areal Nonpoint Source Watershed Environment Response Simulation (ANSWERS), EPIC, European Soil Erosion Model (EUROSEM), Groundwater Loading Effects of Agricultural Management Systems (GLEAMS), Limburg Soil Erosion Model (LISEM), Soil and Water Assessment Tool (SWAT), Griffith University Erosion System Template (GUEST), and Water and Tillage Erosion Model (WATEM). These models have multi-purpose use and can predict not only runoff and soil loss but also nutrient losses. Some models have the ability to simulate subsurface water flow or lateral flow influencing transport of pollutants. Process-based models such as WEPP, SWAT, and AGNPS are particularly popular to simulate impact of contrasting scenarios of land use and tillage and cropping systems on non-point source pollution. Models such as WEPP, LISEM, and EUROSEM simulate soil erosion based on the theory that deposition occurs when concentration of sediment in runoff water surpasses the runoff transport capacity, whereas GUEST estimates erosion based on the simultaneous transport and deposition processes (Yu, 2003).

4.10 Modeling Wind Erosion

Similar to those for water erosion, a number of empirical and physically-based models exist for predicting wind erosion. Models vary in rigor and strictness with which factors and processes are considered. The available models are under continuous refinement to incorporate the complex and variable parameters that govern wind erosion. Most of the current knowledge on the dynamics and mechanics of wind behavior for the development of models comes from the work done in the deserts of North Africa by Bagnold (1935). The first investigations dealt with why and how sand particles accumulated in dunes and what interactive mechanisms occurred between wind blowing and flying soil particles. Climate and soil surface characteristics were recognized as the first drivers of wind erosion early in research. Chepil and Milne (1941) and Chepil (1945a, b, c) expanded the theoretical basis on the mechanics of wind erosion. Chepil (1959) proposed a generalized equation to estimate wind erosion as

$$E = IRKFBWD \tag{4.28}$$

where I is soil cloddiness factor, R is crop residue factor, K is ridge roughness equivalent factor, F is soil abradability factor, B is wind barrier factor, W is width of field factor, and D is wind direction factor. Equation (4.1) estimates field erodibility based on soil erodibility (I and F) and surface erodiblity (R and K).

One of the simplest empirical equations was developed by Pasák (1973) as

$$E = 22.02 - 0.72P - 1.69V + 2.64Rr \tag{4.29}$$

where E is erodibility (kg ha^{-1}), P is percent of non-erodible fraction of soil, V is relative soil moisture, and R_r is wind velocity (km h^{-1}). This model has limited use because does not incorporate variables for vegetative surface cover and soil roughness.

4.11 Wind Erosion Equation (WEQ)

The Wind Erosion Prediction Equation (WEQ) is the classical equation of wind erosion prediction (Woodruff and Siddoway, 1965). It emerged after many years of extensive research on wind erosion dynamics and is represented as

$$E = f(I, K, C, L, V) \tag{4.30}$$

where E is average annual soil loss (Mg ha^{-1} yr^{-1}), I is soil erodibility index (Mg ha^{-1} yr^{-1}), K is soil ridge roughness factor, C is climate factor, L is width of the unsheltered field (m), and V is equivalent vegetative cover factor. The values for the WEQ factors were initially presented in simplified tables and charts (Woodruff and Siddoway, 1965). Later, workable equations were derived from the graphs for computing WEQ factors. The WEQ is the most widely used wind erosion model. While WEQ has limitations for predicting soil erosion rates for a single storm or on a daily basis, it provides useful estimates of wind erosion rates. The WEQ is an empirical model and assumes that wind erosion varies linearly with changes in climate, soil properties, and surface conditions and does not fully account for the complex interactions, combinations, and spatial variability of erosion processes and factors.

4.11.1 Erodiblity Index (I)

The WEQ estimates the potential annual soil loss for a field that is bare, smooth, uncrusted, and well-defined with non-eroding boundaries. The initial I values for the development of WEQ were obtained from field measurements near Garden City, Kansas in the early 1950's using the wind tunnel method (Woodruff and Siddoway, 1965). The WEQ computes I as a function of percentage of >0.84 mm non-erodible aggregates (AGG) near the soil surface determined by the dry sieving method as

$$I = 525 \times (2.718)^{(-0.04AGG)} \tag{4.31}$$

Woodruff and Siddoway (1965) provided a table of I for different percentages (tens) of non-erodible aggregates. The soil erodibility is affected by the presence of knolls in soils with complex topography. Knolls are microrelief features with abrupt windward slopes that increase wind velocity and turbulence. The steeper the slopes of knolls, the greater the wind erosion. The WEQ uses adjustment factors to account for the influence of knolls on I for windward slopes <150 m long (Woodruff and Siddoway, 1965). The erosion rates for windward slopes >150 m are the same as those from flat fields (Woodruff and Siddoway, 1965).

4.11.2 Climatic Factor (C)

The WEQ groups the weather parameters in a climatic factor (C) for estimating soil erosion (Woodruff and Siddoway, 1965; Skidmore, 1986) as

$$C = \frac{1}{100} \sum_{i=1}^{i=12} U^3 \left(\frac{ETP_i - P_i}{ETP_i} \right) d \tag{4.32}$$

where U is mean monthly wind velocity at 2 m height (m s^{-1}), ETP_i is monthly evaporation (mm), P_i is monthly precipitation (mm), and d is number of days in the month of consideration. Equation (4.32) is an index of climatic erosivity in an integrated form. Estimated values of factor C are depicted in iso-C value maps for the region under consideration. The relative potential soil loss for a specific region is determined based on the distribution of C values. In the USA, C values $<10\%$ are for very low, 11–25% for low, 26–80% for intermediate, 81–150% for high, and $>150\%$ for very high relative potential soil loss as a percentage to the potential soil loss near Garden City, Kansas (Chepil et al., 1962).

4.11.3 Soil Ridge Roughness Factor (K)

The ridge roughness (K_r) is estimated by using Eq. (4.33) (Zingg and Woodruff, 1951)

$$K_r = 4\frac{H^2}{S} \tag{4.33}$$

The ridge roughness factor (K) is computed as follows (Woodruff and Siddoway, 1965; Schwab et al., 1993):

$$K = 0.34 + \frac{12}{K_r + 18} + 6.2 \times 10^{-6} K_r^2 \tag{4.34}$$

Skidmore (1983) developed a table of K for different values of ridge height ranging between 25.4 and 254.0 mm and ridge spacing ranging between 25.4 and 1219.2 mm. The K_r is 0 for bare, flat, and smooth fields, so K factor is equal to 1.

Example 3. A recently tilled 800 m long field has abundant well-oriented ridges of 100 mm tall. Estimate the roughness factor if the spacing between ridges is 700 mm.

$$K_r = 4\frac{H^2}{S} = 4\frac{(100\,\text{mm})^2}{700\,\text{mm}} = 57.14\,\text{mm}$$

$$K = 0.34 + \frac{12}{K_r + 18} + 6.2 \times 10^{-6} K_r^2$$

$$= 0.34 + \frac{12}{57.14 + 18} + 6.2 \times 10^{-6}\,(57.14)^2 = 0.52$$

4.11.4 Vegetative Cover Factor (V)

The V is equal to the small grain equivalent $(\mathrm{SG})_\mathrm{e}$ and is estimated as follows (Lyles and Allison, 1981):

$$(SG)_e = aR_w^b \tag{4.35}$$

where $(\mathrm{SG})_e$ is expressed in kg ha^{-1}, a and b are constants (Table 4.4), and R_w is quantity of residue expressed as their small grain equivalent (kg ha^{-1}).

Table 4.4 Coefficient for the prediction of small grain equivalent for selected crops (After Lyles and Allison, 1981)

Crop	Orientation	a	b
Cotton	Flat –random	0.077	1.168
	Standing	0.188	1.145
Silage Corn	Standing	0.229	1.135
Soybeans	1/10 standing	0.016	1.553
	9/10 flat-random	0.167	1.173
Wheat	Flat –random	7.279	0.782
	Standing	4.306	0.970

The WEQ uses two related equations to compute the average annual soil loss based on the computed value of IKCL (Schwab et al., 1993). If the product of IKCL is $> 5.5 \times 10^6$ use Eq. (4.36), otherwise use Eq. (4.37)

$$E = 2.718^{\left(\frac{-V}{4500}\right)} \times \left(I \times K \times \frac{C}{100}\right) \tag{4.36}$$

$$E = 0.0015^{*}2.718^{\left(\frac{-V}{4500}\right)} \times \left[I^{1.87} \times K^2 \times \left(\frac{C}{100}\right)^{1.3} L^{0.3}\right] \tag{4.37}$$

Example 4. Estimate the amount of soil lost from the field in Example 3 which has 20% of non-erodible knolls, and 2 Mg ha^{-1} of residue remaining on the soil surface under corn silage. The region is under a high potential soil loss with factor C value of 85%.

First, compute I and V

$$I = 525\,(2.718)^{-0.04\times 20} = 235.917\,\text{Mg ha}^{-1}$$
$$V = aR_w^b = 0.229(2000)^{1.135} = 1277.92\,\text{Kg ha}^{-1}$$

Then, compute and check IKCL product

$$IKCL = 235.917 \times 0.652 \times 85 \times 800 = 10459616.11$$

Since IKLC is $> 5.5 \times 10^6$ use

$$E = 2.718^{\left(\frac{-V}{4500}\right)} \times \left(I \times K \times \frac{C}{100}\right)$$

$$E = 2.718^{\left(\frac{-1277.92}{4500}\right)} \times \left(235.917 \times 0.652 \times \frac{85}{100}\right) = 98.43\,\text{Mg ha}^{-1}\,\text{yr}^{-1}$$

Example 5. Estimate the soil loss if the length of field in Example 4 decreases to 100 m and the factor C value to 10%.

$$IKCL = 235.917 \times 0.652 \times 10 \times 100 = 153817.88$$

Since IKLC is $< 5.5 \times 10^6$ use

$$E = 2.718^{\left(\frac{-V}{4500}\right)} \times \left[I^{1.87} \times K^2 \times \left(\frac{C}{100}\right)^{1.3} L^{0.3}\right] = 2.59\,\text{Mg ha}^{-1}$$

The reduction in C value and field length reduced soil loss by about 38 times.

4.12 Revised WEQ (RWEQ)

Similar to USLE, the WEQ has also undergone an extensive revision since it was developed in the 1960's. What started as an empirical equation has become a highly sophisticated model through continuous refinement. Improvement in WEQ model led to the emergence of the RWEQ in 1998 (Fryrear et al., 1998), which is a more structured model and portrays better the physical processes of wind erosion. It combines extensive field data sets with computer models to assess soil erosion at local and regional scales. The RWEQ estimates erosion based on wind velocity, rainfall characteristics, soil roughness, erodible fraction of soil, crusts, amount of surface residues, and other dynamic parameters. It predicts mass transport of soil by wind based on weather factor (WF), erodible fraction of the soil (EF), soil crust factor (SCF), soil roughness factor (K'), and combined crop factors (COG).

The RWEQ estimates horizontal mass transport using the steady state equation as a basic principle

$$b(x)\frac{dQ(x)}{dx} + Q(x) - Q_{\max}(x) + S_r(x) = 0 \qquad (4.38)$$

where $b(x)$ is field length (m) and varies with length of field, Q_x is maximum amount of soil transported by wind at field length x, downwind distance, at a height of 2 m (kg m^{-1}), Q_{max} is maximum transport capacity over that field surface (kg m^{-1}), x is total field length (m), and S_r is surface retention coefficient. Assuming $Q_{\max}$ and b are constants on a uniform field. Eq. (4.38) is simplified into a sigmoidal form to estimate the downwind transport of soil through a point x as

$$Q_x = Q_{\max}\left[1 - e^{-\left(\frac{x}{s}\right)^2}\right] \qquad (4.39)$$

where s is critical field length at which $Q_{(s)}$ is equal to 63% of $Q_{\max}$. The $Q_{\max}$ and s are estimated as

$$Q_{\max} = 109.85(WF \times K \times EF \times SCF \times COG) \qquad (4.40)$$

$$s = 150.71\,(WF \times EF \times SCF \times K \times COG)^{-0.3711} \qquad (4.41)$$

4.12.1 *Weather Factor (WF)*

The WF is a function of wind, snow, and soil wetness and is estimated as

$$WF = W_f\frac{\rho}{g}(SW)\,SD \qquad (4.42)$$

$$W_f = \frac{W}{500} \times N_d \tag{4.43}$$

$$SW = \frac{ET_p - (R + I)\,\frac{R_d}{N_d}}{ET_p} \tag{4.44}$$

$$ET_p = 0.0162\left(\frac{SR}{58.5}\right)(DT + 17.8) \tag{4.45}$$

$$W = \sum_{i=1}^{i-n} U_2\,(U_2 - U_t)^2 \tag{4.46}$$

where W_f is wind factor (m^3 s^{-3}), W is wind value (m^3 s^{-3}), N_d is number of days in the study period, *SW* is soil wetness factor, ET_p is potential relative evapotranspiration (mm), R_d is number of rainfall and/or irrigation days, R is rainfall amount (mm), I is cumulative infiltration (mm), *SR* is total solar radiation for the study period (cal cm^{-2}), *DT* is average temperature (˚C), U_2 is wind velocity at 2 m (m s^{-1}), and U_t is wind velocity at 2 m equal to 5 m s^{-1}.

4.12.2 Soil Roughness Factor (K)

The K is computed (Fryrear et al., 1998) as

$$K = \exp^{\left(1.86 K_r R_c - 2.41 K_r R_c^{0.934} - 0.124 C_{rr}\right)} \tag{4.47}$$

$$R_c = 1 - 3.2 \times 10^{-4}\,(A) - 3.49 \times 10^{-4}\left(A^2\right) + 2.58 \times 10^{-6}\left(A^3\right) \tag{4.48}$$

$$C_{rr} = 17.46 RR^{0.738} \tag{4.49}$$

where R_c is wind angle assumed 0 degrees for perpendicular and 90 degrees for parallel angles and *RR* is random roughness index (Allmaras et al., 1966). Saleh (1993) developed an alternative form to compute K_r as

$$K_{r_{(ch\,\sin)}} = \frac{0.08 H^2 W R}{S[(2W) - S]} \tag{4.50}$$

where H is ridge height (m), S is ridge spacing (m), W is side of an isosceles ridge (m), and R is surface roughness index (%) equal to

$$R = \left(1.0 - \frac{S}{2W}\right) 100 \tag{4.51}$$

4.12.3 Erodible Fraction (EF)

The EF in the RWEQ is estimated as

$$EF = \frac{29.09 + 0.31Sa + 0.17Si + 0.33\frac{Sa}{Cl} - 2.59SOM - 0.95CaCO_3}{100} \tag{4.52}$$

where *Sa* is sand content (%), *Si* is silt content (%), *Sa/Cl* is sand to clay ratio, *SOM* is soil organic matter content (%), and $CaCO_3$ content (%).

4.12.4 Surface Crust Factor (SCF)

The empirical relationship to estimate SCF is

$$SCF = \frac{1}{\left[1 + 0.0066\,(clay)^2 + 0.21\,(SOM)^2\right]} \tag{4.53}$$

4.12.5 Combined Crop Factors (COG)

The COG simulates the effect of crop canopies, plant silhouette, and standing and flat residues on erosion using equations developed from lab wind tunnel experiments. The COG specifically characterizes the fraction of land covered by plant materials by multiplying soil loss ratio for cover (SLR_f), plant silhouette (SLR_s), and growing canopy cover (SLR_c).

$$COG = SLR_f \times SLR_s \times SLR_c \tag{4.54}$$

$$SLR_f = \exp\,[-0.0438\,(SC)] \tag{4.55}$$

$$SLR_s = \exp\,[-0.0344\,(SA)] \tag{4.56}$$

$$SLR_c = \exp\left[-5.614\left(cc^{0.7366}\right)\right] \tag{4.57}$$

where *cc* is fraction of soil surface covered by crop canopy. Finally, the average soil loss (SL) in kg m^{-2} for a specific field of length x is computed as

$$SL = \frac{Q_{\max}}{x} \tag{4.58}$$

While REWQ is better than WEQ in terms of flexibility of input parameters, it still shows some limitations to accurately predict erosion for: (1) within-field conditions, (2) fields without non-eroding boundaries, and (3) transport of particles in suspension (e.g., dust emissions). Daily changes in soil roughness and freezing/thawing as result of fluctuations in weather and management are not simulated by RWEQ. The RWEQ combines empirical and process-based approaches for the prediction, and thus it is not completely a physically-based model.

4.13 Process-Based Models

A simple but physically-based model is that derived by Stout (1990). This equation is an exponential curve that simulates transport mass of soil loss per unit area as follows:

$$Q_x = Q_{\max}\left[1 - \exp^{-\left(\frac{x}{s}\right)^2}\right] \quad (4.59)$$

where Q_x is mass of soil transported by wind at field length x in kg m^{-1}, Q_{max} is maximum transport capacity in kg m^{-1}, x is field length in m, and s is inflection point where slope of curve switches from positive to negative.

4.14 Wind Erosion Prediction System (WEPS)

The WEPS is a new prediction technology and was designed to replace WEQ because it is a process-based, continuous, daily time-step, and computer-based model. It differs from WEQ because it simulates wind erosion based on physically-based processes of erosion (Table 4.5). The WEPS provides better estimates of wind erosion than other models (Visser et al., 2005). It simulates complex field conditions

Table 4.5 Differences between WEQ and WEPS

WEQ	WEPS
• Uses empirical parameters	• Uses process-based modeling parameters
• Predicts erosion for a single and uniform field. Fields with high spatial variability are treated like uniform fields.	• Predicts erosion for nonuniform fields. It partitions a spatially variable field in subfields with similar topographic characteristics.
• Predicts average erosion across the field and treats it as uni-dimensional.	• Treats a field as two-dimensional by simulating erosion for each grid point. It models saltation/creep separate from suspension.
• Predicts only long-term and average soil loss.	• Predicts erosion for single storms on a daily, weekly, monthly, and yearly basis.
• Simulates no interactive erosion processes and relies solely on the input of parameters by users.	• Simulates a whole range of wind and intrinsic soil properties and processes in relation to soil surface and management conditions.
• Neglects influences of daily or periodic cyclical weather fluctuations (e.g., rainfall) on erosion.	• Accounts for the periodic interactive effects among climate, soil properties, vegetation, surface roughness, and management.

accounting for the spatial and temporal variability, and it separately simulates transport processes of suspension, saltation, and creep. It relies heavily on the dynamics of soil properties and processes and can estimate wind erosion damage to crops and determine air pollution with dust emissions (PM-10 and PM2.5). The WEPS combines complex set of mathematical equations to predict erosion (Hagen, 1996).

The structure of WEPS consist of a MAIN routine, seven submodels (weather, hydrology, soil, crop, decomposition, management, and erosion), and four databases (climate, soils, management, and crop) (Hagen, 1996) (Table 4.6). The WEPS is specifically designed to assist land managers and extension agents in understanding processes of soil erosion and controlling wind erosion from croplands, forestlands, rangelands, pasturelands, and any disturbed (e.g., construction sites) land. It is still under refinement for handling topographically complex terrains and hydrologically diverse soils under different regions. Training tools for using the WEPS model are well documented (Hagen, 1996; USDA, 2006).

Specific recent improvements in WEPS include changes in the java codes and user interface with multiple WEPS run in the same window, new-updated on-line user's guide, incorporation of a new submodel "WEPP-based hydrology/infiltration/evaporation", expansion of data on wind characteristics and command options for irrigation practices, estimation of erosion for fields of different shape, and development of a "Single-event Wind Erosion Evaluation Program" (SWEEP) (USDA, 2006).

Table 4.6 Structure of WEPS model

Submodel	Simulates
WHEATER	• Wind characteristics (e.g., intensity, direction, friction velocities)
HYDROLOGY	• Soil temperature • Soil water content (e.g., snow melt, runoff, infiltration, deep percolation, evaporation , evapotranspiration rates)
SOIL	• Processes (e.g., wetting-drying, freezing-thawing) • Properties (e.g., bulk density, aggregate density, aggregate size distribution, surface roughness, and thickness, strength, stability of crusts)
CROP	• Wind erosion effects on plant growth • Changes in vegetative cover
DECOMPOSITION	• Decomposition rates of plant residues production of leaves, stems, and roots
EROSION	• Particle transport processes based on the conservation of mass in relation to surface cover and roughness. • Effect of windbreaks, field borders, and buffers • Changes in topographic conditions within the same field.
MANAGEMENT	• Diverse cultural and management practices • Primary and secondary tillage, fertilization, residue management, manuring, seeding, irrigation, harvesting, grazing, and burning of residues.

4.15 Other Wind Erosion Models

4.15.1 Wind Erosion Stochastic Simulator (WESS)

The WESS is a process-based model that has the ability to simulate wind erosion on an event basis (Potter et al., 1998). It uses soil texture, erodible soil thickness, bulk density, erodible particle diameter, soil roughness, soil water content, amount of crop residue, 10 min average wind velocities, and field size as input. The WESS is a module of the EPIC model. Estimates of soil erosion by WESS when compared to those by other models are promising (Van Pelt et al., 2004).

4.15.2 Texas Tech Erosion Analysis Model (TEAM)

The TEAM is also a process-based model with an ability to simulate movement of particles in suspension and saltation. It uses wind velocity and distribution, relative humidity, soil roughness, and particle size distribution as main input parameters. The TEAM has been used in sandy soils under agricultural and industrial uses. Information from the TEAM was used to develop strategies for stabilizing moving sand dunes in desert regions (Gregory et al., 2004).

4.15.3 Wind Erosion Assessment Model (WEAM)

The WEAM is a physically-based model that simulates sand entrainment from different sites (Shao et al., 1996). It is based on the Owen equation for simulating saltation flux and dust entrainment. Particle size distribution is the primary input for the model. The WEAM is most useful to describe sand particle entrainment and not as much for dust transport and deposition.

4.15.4 Wind Erosion and European Light Soils (WEELS)

The WEELS is a spatially distributed erosion model under development that predicts erosion rates at different time scales (Böhner et al., 2003). It is structured in a way that it simulates different cropping, management, and climatic scenarios. The WEELS consists of six modules such as wind, wind erosivity, soil water, soil erodibility, soil roughness, and land use. These modules simulate the temporal variations of wind, soil, and vegetation cover characteristics. The WEELS has limitations for simulating soil water dynamics for sandy soils and its use is mostly restricted to fine textured soils. It can not simulate the sediment flux in suspension or dust emissions. Characterization of net soil loss using this model is mostly based on sediment particles in saltation.

4.15.5 Dust Production Model (DPM)

The DPM combines processes of saltation and sandblasting to estimate the amount of aerosol (Alfaro and Gomes, 2001). It is based on the principle that wind velocity, dry size distribution of the soil aggregates, and roughness length define the release of $<20\,\mu\text{m}$ dust particles.

4.16 Limitations of Water and Wind Models

While the available models have advanced the understanding of soil erosion processes and estimation of erosion rates, their applicability to conditions different from those for which they were developed remains limited. The large and detailed data required as input for most of the current models are seldom available. Performance of current models is highly variable and site-specific. Further model development of process-based models for a wide range of soil, management and climate scenarios is warranted. Model domains must incorporate the temporal and spatial variability of conditions.

Summary

Modeling water and wind erosion is essential to the understanding of processes and estimating rates of soil erosion. The estimates are needed to design and implement erosion control measures. Modeling is also useful to scaling up of information from small-scale experiments to larger geographic areas and estimate soil erosion on a regional and national basis. Empirical (e.g., USLE) and process-based (e.g., WEPP) erosion models are available for modeling soil erosion. Compared to process-based models, empirical equations such as the USLE require fewer input parameters and are thus more adaptable to scenarios with limited database. The USLE does not, however, simulate the soil detachment, transport, and deposition processes and is designed to predict soil loss from small plots. The MUSLE and RUSLE are the result of the ongoing development of the empirical models.

The process-based models are based on the sediment continuity equation. The WEPP is a common a process-based model that estimates erosion for single hillslopes and whole watersheds on a temporal (daily, monthly, and annual basis) and spatial (hillslopes and small, medium, and large watersheds) scales. It integrates information on weather conditions, tillage and soil management, soil hydrology, plant growth, soil parameters, erosion and deposition, channel hydrology and erosion processes, and watershed processes. The EGEM is another process-based model, which is designed to predict gully erosion on an event basis.

The WEQ is a common empirical equation to predict wind erosion. Its input parameters are soil erodibility, soil roughness, climate, field width, and vegetative cover. The RWEQ, a revised form of WEQ, is a more structured model than WEQ and incorporates wind velocity, rainfall characteristics, soil roughness, erodible fraction of soil, crusts, amount of surface residues, and other dynamics parameters for predicting erosion. The WEPS is a process-based and continuous model unlike WEQ. It uses process-based parameters to predict erosion for single storms on a daily, weekly, monthly, and yearly basis. It simulates the interactive effects among climate, soil properties, vegetation, surface roughness, and management.

Current erosion models have limitations to accurately predict soil erosion. The large database required as input hampers the applicability of some models. Available models are highly variable and site-specific. Development of a comprehensive and unique water and wind erosion model is needed. Combination of current models with advanced tools such as remote sensing and GIS is a promising approach to enhancing the predictive ability of models at different temporal and spatial scales.

Study Questions

1. Estimate the kinetic energy of a rainstorm if the average rainfall amount of 30-min duration is 50 mm of constant intensity.
2. Estimate the average annual soil loss for 300 m field, moldboard plowed under corn-soybean rotation in mid Missouri. The soil is silt loam (17% coarse and medium sand, 3% very fine sand, 22% clay, and 58% silt) with 2.1% of soil organic matter content and slope of 3.8%. The structure is fine granular and the saturated hydraulic conductivity is 55 mm h^{-1}.
3. Predict the K_b for a soil with 30% of clay content and 15% of sand content. The CE is 1.5 meq $(100\,g)^{-1}$. What is the magnitude of K_b increase if the clay content increases to 45%.?
4. Compute the erodible fraction and surface crust factor for the soil in Prob. 2 if the mean $CaCO_3$ content is 2.5%.
5. Discuss and compare the input parameters for USLE and WEPP.
6. Describe each term of the sediment continuity equation.
7. What are the differences between empirical and physically-based models.
8. What are the similarities and differences between WEQ and WEPS models.
9. Discuss the shortcomings of current models.
10. Explain the reasons as to why empirical models of erosion are more widely used than process-based models.

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Chapter 5
Tillage Erosion

Tillage is the "mechanical manipulation of the soil for any purpose" (SSSA, 2007). It is important to seedbed preparation, weed control, and incorporation of agricultural chemicals or amendments. Although no-till management is generally preferred over practices that disturb soil, an occasional tillage may be necessary, in some soils (e.g. clayey), to: (1) alleviate excessive soil compaction, (2) reduce interference of surface crop residues during plant establishment, and (3) offset stratification of nutrients and soil organic matter due to the confinement of crop residues to the soil surface. Certain no-till planters have attachments to slightly loosen the soil and remove crop residues from the rows while placing seeds. Major concern with tillage arises when it becomes intensive and continuous, which drastically alters soil functions and cause soil erosion. Intensive tillage operations destroy the natural soil structure, overturn and drag the loose material, and redistribute soil downhill along the lower landscape positions (Fig. 5.1).

Fig. 5.1 Plowing shifts soil downhill (Courtesy T.E. Schumacher, South Dakota State Univ.)

5.1 Definition and Magnitude of the Problem

Tillage erosion refers to the gradual soil translocation or displacement downhill caused by tillage operations (Lindstrom et al., 1990). It is also known as "dry mechanical erosion" because it refers to the erosion of soil by mechanical manipulation without the water action. The net soil translocation by tillage is expressed in units of volume, mass, or depth per unit of tillage width. Traditionally, water, wind, and gravity erosion have been considered the only drivers of total soil erosion. In recent years, particularly with the advent of mechanized agriculture, tillage erosion has become an important component of total soil erosion in hilly croplands. Soil erosion by tillage can be extremely high and hence not "sustainable". Between 15 and 600 Mg ha^{-1} of soil can be lost by tillage erosion annually in hilly croplands (Table 5.1). Tillage erosion can represent as much as 70% of total soil erosion (Lobb et al., 1999).

Tillage erosion is a serious soil degradation process in sloping cultivated soils worldwide. Agriculture in humid and semi-arid tropics is normally practiced on steep slopes. As a result, large areas which have been recently incorporated into croplands are being abandoned due to severe tillage erosion (St. Gerontidis et al., 2001). Tillage erosion modifies the landscape geomorphology by progressively removing topsoil layers from convex field positions (summits, crests, and shoulder slopes) and redistributing the removed materials along concave landscape positions (foot- and toe-slopes) (Fig. 5.2). Tillage erosion modifies the spatial patterns of landform elements while inducing changes in soil properties. It negatively impacts fluxes of water, heat, and air, soil-water-plant relations, nutrient cycling and availability, and other dynamic processes. While soil erosion by water and wind is still the predominant mechanism of soil erosion, tillage is also a critical cause of soil erosion in sloping landscapes. It is a continuous process where the convex landscapes positions gradually lose soil while the concave areas aggrade soil (Fig. 5.2). The slow but gradual removal of topsoil by tillage exposes subsoil and thereby jeopardizes the overall soil productivity and environmental quality.

Table 5.1 Rates of tillage erosion for selected regions

Country	Tillage Direction	Soil Slope (%)	Tillage Method	Soil Erosion (Mg ha^{-1} yr^{-1})
China[1]	Downslope	43	Manual hoeing	48–151
Spain[2]	Downslope	40	Moldboard plow	68
China[3]	Downslope	40	Animal traction	22
Portugal[4]	Downslope	25	Moldboard plow	35
Philippines[5]	Up- and down-slope	45	Moldboard plow	456–601
Belgium[6]	Up- and down-slope	14	Moldboard and chisel plow	10
USA[7]	Up- and down-slope	14	Moldboard plow	30

[1]Zhang et al. (2004a), [2]De Alba (2003), [3]Li and Lindstrom et al. (2001), [4]Van Muysen et al. (1999), [5]Thapa et al. (1999), [6]Govers et al. (1994), and [7]Lindstrom et al. (1992).

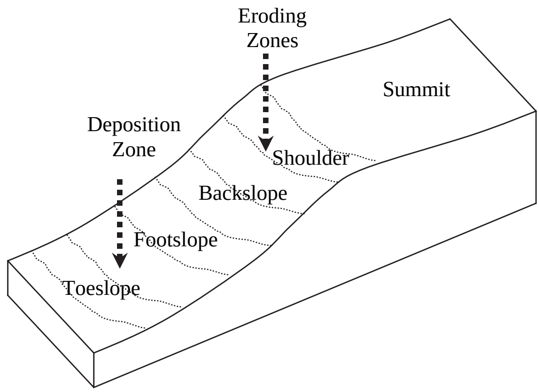

Fig. 5.2 Hillslope profile characteristics as factors for gains and losses of soil by tillage

5.2 Tillage Erosion Research: Past and Present

Tillage erosion has been recognized since 1930s, yet it is an emerging field of research in terms of the processes involved. In the USA, three pioneering studies by Nichols and Reed (1934), Mech and Free (1942), and Chase (1942) described implications of plow tillage for displacing soil downslope in sloping croplands. Between 1950s and 1980s, only a few researchers specifically studied the tillage erosion. Most research focused on the mechanics of soil shearing, deformation, and "flow" in contact with tillage tools (Fornstrom et al., 1970). In the Pacific Northwest, dramatic signatures of tillage erosion causing formation of soil banks of 3–4 m high along the field boundaries were reported (Papendick and Miller, 1977).

Detailed and quantitative studies on tillage erosion were uncommon until the early 1990s when Lindstrom et al. (1990; 1992) conducted comprehensive assessments of soil translocation in the USA and reported that soil displacement from convex to concave field points due to tillage was far from insignificant. These works sparked a heightened interest in tillage erosion research. Tillage erosion is now regarded as one of the most important soil degradation processes on sloping agricultural soils. This recognition has caused a rapid shift in soil erosion research since mid 1990s.

Introduction of heavy and aggressive tillage equipment favors the plowing of extensive areas in a short time period, dramatically increasing soil's susceptibility to erosion by tillage, water, and wind across the world, especially in countries with highly mechanized agriculture. Thus, tillage erosion is the greatest risk under moldboard plowing. Concerns over increased tillage erosion with mechanized implements have also stirred interest in assessing implications of animal-pulled implements or manual tillage on soil erosion.

The current trend in tillage erosion research is to unravel complex interactions among differing scenarios of slope gradients, soil conditions, and tillage operations (e.g., tillage direction, speed, soil depth) that affect the magnitude of tillage erosion under either controlled or field management systems. Computer modeling, GIS tools, and statistical modeling are now used for studying magnitude and ramifications of tillage erosion across different ecosystems. Computer modeling allows the prediction of tillage erosion and enhances the understanding of the erosional processes.

5.3 Tillage Erosion versus Water and Wind Erosion

Soil erosion by tillage differs from that by water and wind with respect to the landscape dynamics (Table 5.2). Water and wind erosion are controlled by rainfall and wind intensity, respectively, while tillage erosion is influenced by the tillage intensity. Wind erosion is the most serious on the shoulder slopes facing the dominant wind direction (Schumacher et al., 1999). Tillage operations, unlike water and wind erosion, rarely transport soil off-site but redistribute it within the field where large amounts of loose soil gradually slump to the toeslope. Both water and tillage erosion strongly depend on slope gradient and soil surface conditions. Tillage erosion removes fertile topsoil and reduces soil productivity. Similar to water and wind erosion, effects of tillage erosion on soil function, crop production, and environmental quality can also be as negative or adverse.

Tillage erosion preferentially removes soil from the shoulder slopes while water erosion mostly transports that from the backslopes. While water erosion often

Table 5.2 Similarities and dissimilarities among tillage, water, and wind erosion

Tillage erosion	Water erosion	Wind erosion
• Depends on tillage intensity	• Depends on rainfall or runoff intensity (erosivity).	• Depends on wind intensity.
• Does not develop rills or gullies	• Creates dissected landscapes by forming rills and gullies.	• Tends to flatten the soil landscape and form dunes.
• Moves soil by sliding/rolling.	• Moves large soil particles by rolling and small particles in suspension.	• Moves soil by saltation and in suspension.
• Moves soil a short distance.	• Can move soil a long distance.	• Can move soil a long distance.
• Does not transport soil off-site.	• Transports the soil off-site.	• Transports the soil off-site.
• Does not develop obvious signs and is a slow process.	• Creates visible signs (rills and gullies) and is a rapid process.	• Develops visible marks and is a rapid process.
• Depends on both slope gradient and soil surface conditions.	• Depends on both slope gradient and length and soil surface conditions.	• Depends mostly on soil surface conditions.
• Inverts and shifts large amounts of soil.	• Erodes thin films of soil.	• Erodes exposed soils.

develops rills, ephemeral, and permanent gullies, creating dissected landscapes, tillage erosion tends to flatten the landscape by translocating soil from convex positions and infilling lower positions. Water erosion events create visible pathways, whereas tillage erosion events are often not as obvious because tillage erosion is a slow process. This may be the reason why research on tillage erosion has received less attention than that by water and wind.

Magnitude of tillage erosion on convex agricultural sloping fields can be equal to or even higher than that by water erosion depending on the tillage direction and site-specific conditions of soil. Rates of soil erosion by tillage (60 Mg ha^{-1} yr^{-1}) can be as much as 3 times higher than those by water erosion (20 Mg ha^{-1} yr^{-1}) as in rolling soils in Minnesota (Papiernik et al., 2005). Unlike water and wind erosion, tillage loosens and inverts the entire plow layer. Tillage operations not only translocate soil but also disperse it, increasing its susceptibility to water and wind erosion. Thus, tillage erosion aside from physically translocating the soil material downslope, indirectly contributes to water and wind erosion. Interactions among tillage, water, and wind erosion processes cause the greatest losses of soil from croplands. It is typical to observe clouds of dust over plowed soils upon drying as a sign of wind erosion. The processes such as removal, transport, and deposition among tillage, water, and wind erosion are similar. Sheet or interrill erosion gradually removes thin layers of surface soil whereas a single tillage pass can cause the downslope movement of an entire slice of plow layer although at shorter distances per plowing event compared to water and wind erosion. While tillage erosion moves soil at short intervals per passage of a plow, it can move equal or more soil than water erosion over time.

5.4 Factors Affecting Tillage Erosion

Establishing the cause–effect relationships of tillage erosion is fundamental to understanding the mechanisms leading to soil translocation in agricultural systems. A number of interactive factors including types and operations of tillage equipment,

Table 5.3 Factors affecting soil displacement over hillslopes by tillage erosion

Landform erodibility	Soil erodibility	Tillage operations	Tillage implements
Slope: • Gradient • Aspect • Length • Shape	• Water content • Soil texture • Gravel content • Stoniness • Bulk density • Cone index • Shear strength • Plant roots • Organic matter content	Tillage: • Depth • Direction • Speed • Number of passes Tool: • Type • Spacing • Width • Orientation	• Moldboard plow • Chisel plow • Disk plow • Animal-drawn tools • Manual tools

landscape morphology, and soil intrinsic characteristics dictate the amount of soil translocated by tillage (Table 5.3). The factors are intrinsic to *tillage erosivity* and *landscape erodibility*.

- ***Tillage erosivity*** refers to the capacity of tillage operations to erode the soil (Van Muysen et al., 2006).
- ***Landscape erodibility*** refers to the susceptibility of a cultivated soil to be eroded by tillage operations (Lobb and Kachanoski, 1999). It comprises both landform and soil erodibility.

$$\text{Tillage Erosion} = f(\text{tillage erosivity, landform erodibility, and soil erodibility})$$

5.5 Landform Erodibility

Magnitude of tillage erosion is correlated with landscape characteristics. Topographically complex terrains are more prone to tillage erosion than relatively flat or uniform lands. Net soil erosion by tillage erosion increases with increase in slope gradient regardless of tillage type. Soil erosion by tillage is often the most serious on the shoulder slopes because of their steep gradient. Translocated soil is redistributed along the hillslope but eventually deposited on foot- and toe-slopes. In contrast to water erosion, tillage erosion does not depend as much on slope length.

5.6 Soil Erodibility

Magnitude of tillage erosion is influenced by the antecedent soil conditions just prior to or at the time of tillage operation. Soil transport by tillage is often higher in unconsolidated than in consolidated soils. Thus, pre-tilled soils are translocated more easily than untilled soils. An increase in soil bulk density results in higher rates of erosion by tillage. Bulk density is a dynamic soil property which changes with soil water content, tillage type, soil texture, and organic matter content. Impacts of soil water and organic matter content on tillage erosion depend on soil textural classes.

5.7 Tillage Erosivity

5.7.1 Tillage Depth

The greater is the depth of tillage implement penetration, the more is the amount of soil available for translocation by tillage operations. In recent years, farmers using mechanized agriculture have increased the plowing depth from 20 to 30 cm, and, in some cases, to even 50 cm. Deep plowing is often used to break plow pans caused by previous tillage or naturally compact layers (e.g., fragipan, claypan, hardpan)

Fig. 5.3 Tillage erosion increases with tillage intensity and slope gradient (Courtesy T.E. Schumacher, South Dakota State Univ.)

(Fig. 5.3). Sub-soiling causes greater soil inversion and destruction of natural soil structure. An increase in tillage depth from 20 to 40 cm can increase soil displacement by 75% (St. Gerontidis et al., 2001). Tillage depth changes during tillage as a function of changes in topography, tillage implement (e.g., flexibility), soil condition (e.g., gravel content, stoniness, soil consolidation), and consistency of the operator. Tillage depth using moldboard plow, chisel plow, tandem disc, and field cultivator can increase by about 20% and decrease by about 30% during plowing in undulating soils as compared to that in flat terrains (Lobb et al., 1999).

5.7.2 Tillage Implement

The amount of soil displaced by tillage is correlated with the type and characteristics of the implement used. Moldboard plow, a widely used tillage technique, is the most erosive tillage implement. Moldboard plow not only loosens and overturns the soil, but it shifts the entire plow layer downslope. It displaces soil more than chisel, disk, or harrow per pass. Width, spacing, orientation, and arrangement of each individual tillage tool influence soil translocation. For example, by turning the furrows upslope, moldboard plowing can minimize soil transport downhill.

Other tillage techniques including animal- and human-powered implements (e.g., hoes) also contribute to tillage erosion in hillslope farming. Downslope non-mechanized tillage can translocate as much soil as the mechanized tillage. Indeed, fields are normally tilled downslope by animal force to reduce the energy spent in tillage. Tillage, using manual implements, is conventionally performed from bottom to top on hilly lands. Thus, these practices translocate loose soil downhill as opposed to mechanized tillage in which upslope tillage may somewhat offset downslope translocation. Manual tillage can be as erosive as mechanized tillage or even higher in steep slopes. An experiment conducted in Tanzania showed that tillage erosion by manual hoeing explained the shallow topsoil in mountainous regions with steep slopes ($>$50%) (Kimaro et al., 2005). In Thailand, some soils with 70 and 80% slopes have lost the entire shallow plow layers by manual tillage (Turkelboom et al., 1999). Manual tillage implements have been used since the dawn of agriculture and are still being widely used in developing countries.

5.7.3 Tillage Direction

Spatial patterns of soil displacement depend on the direction of tillage. The maximum soil displacement occurs when tillage is performed in a downslope direction. This type of tillage in interaction with gravity causes rapid sliding and rolling of the plowed layer. When tillage is performed in an upslope direction, soil is moved upward and displacement is reduced, but it can be counteracted by the gravity that pushes the plowed material downhill. Up- and down-slope tillage results in a net downslope translocation of soils in response to gravity. Gravity can readily overcome upslope tillage in steep slopes. Interaction of slope gradient with tillage direction defines the maximum downslope soil translocation (De Alba et al., 2006). Maximum downhill soil displacement using a moldboard plow occurs at a tillage direction of 60 and 70° and not at 0° when tractor moving downhill is defined at 0° and uphill at 180° (Torri and Borselli, 2002). The parallel distribution of the soil to the direction of tillage creates forward translocation while the perpendicular distribution creates lateral translocation.

Contour tillage is the most preferred technique to minimize tillage erosion. It can reduce erosion rates by 75 and 85% compared to downslope tillage (Zhang et al., 2004b). Soil displacement can be reduced by 70–95 cm by changing the plowing direction from downslope to contour for an equal tillage depth (St. Gerontidis et al., 2001). Soil transported by downslope tillage can be twice as much as that by contour tillage for moldboard plow. Similarly, soil displacement in upslope tillage can be twice lower than that in the downslope tillage direction (St. Gerontidis et al., 2001). Soil transport increases exponentially when tillage is perpendicular to the contour lines and linearly than when it is parallel to the contour lines.

5.7.4 Tillage Speed

Tillage speed is the principal control of soil displacement and transport, which increases linearly with increase of the tractor speed. It is estimated that a reduction of tillage speed from 7 to 4 km h^{-1} reduces tillage erosivity by about 30% (Quine et al., 2003). The expansion of agriculture has favored the use of high tillage speeds to cover large areas, resulting in intensification of tillage erosion. The preset tractor speed changes during tillage, depending on the landscape heterogeneity and soil characteristics. The tractor speed can decrease by about 60% during upslope tillage and increase by about 30% during downslope tillage (Lobb et al., 1999).

5.7.5 Frequency of Tillage Passes

The higher is the number of implement passes, the larger is the amount of soil displaced. In humid regions with bimodal rains, soil is normally plowed twice a year, causing more displacement than single plowing. In tropical, semi-arid, and arid regions, soil is mostly plowed once annually and is also accompanied by hoeing.

5.8 Tillage Erosion and Soil Properties

Soil displacement by tillage causes dramatic changes in soil profile characteristics and soil properties (Table 5.4):

5.8.1 Soil Profile Characteristics

Tillage erosion truncates soil profiles on the shoulderslopes and modifies the soil profiles downslope. It affects soil formation and horizonation. Thickness of the A horizon decreases significantly as the soil is eroded. The A horizons are shallower on the shoulder slopes and thicker on the footslopes (Fig. 5.4). The A horizon on the footslopes can be as thick as 50 cm. In Minnesota, annual moldboard plowing for 40 yr exposed the calcareous subsoil horizon in convex zones, and thus increased the depth of the A horizons in the depositional areas (De Alba et al., 2004). The original topsoil in the shoulder-slope positions is replaced by the subsoil while that in the

Table 5.4 Tillage erosion impacts on soil properties of the eroding sites

Physical properties	Chemical and biological properties
Increase in:	***Decrease in:***
• clay content	• organic matter content
• bulk density	• nutrient content
• penetration resistance	• CEC
• gravel content	• base saturation
Decrease in:	• proliferation and activity of macro-, and micro-organisms
• water transmission rates	• above- and below-ground biomass
• air and heat fluxes	
• aggregation	

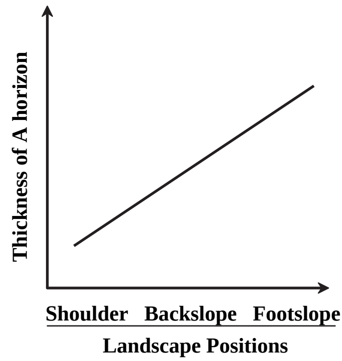

Fig. 5.4 Changes in A horizon thickness due to accelerated tillage erosion

foot- and toe-slope positions is eventually buried with the upstream translocated soils.

5.8.2 Soil Properties

Changes in profile characteristics concomitantly affect within-field variability of soil properties of the topsoil and the underlying horizons. Removal of topsoil from the shoulder positions exposes subsoil of different physical, chemical, and biological characteristics from the original soil. Soil deposition in concave areas also modifies the properties of the underlying horizons. Tillage erosion alters soil properties at both the eroding and aggrading sites. Soil texture, bulk density, porosity, water retention capacity, hydraulic conductivity, organic and inorganic C pool, pH, and biological activities are among the first soil properties readily altered by tillage erosion (Table 5.4). Shoulder slopes normally have higher clay content than footslopes due to the exposure of clay-enriched subsoil layers by tillage erosion (Heckrath et al., 2005). The exposure of clayey subsoil negatively impacts soil structural development, water retention, and nutrient cycling. The exposed sub-soils are rich in carbonate content but poor in soil organic C content (Papiernik et al., 2005). Shallow soils on the convex positions have also higher rates of evaporation and drainage than those on concave positions, which have deeper soils and higher soil water content. Losses in organic matter are linearly correlated with those of soil (Li and Lindstrom, 2001). Tillage erosion also translocates nutrients and chemicals to low lying areas, a process that may cause non-point source pollution.

5.9 Indicators of Tillage Erosion

Changes in soil properties, soil surface elevation, and spatial distribution of radionuclides along hillslopes are used to assess the rate and magnitude of tillage erosion. As discussed previously, tillage erosion alters soil properties, which can thus be used as indicators of occurrence of erosion.

5.9.1 Changes in Surface Elevation

Changes in surface elevation at both eroding and aggrading sites are likely indicators of the recurrent and intensive tillage erosion. The surface elevation of eroding sites decreases while that of aggrading sites increases under significant tillage erosion. Soil denudation rates due to chisel plow can be as high as 3.6–5.9 mm yr^{-1} for up- and downslope tillage and up to 1.5–2.6 mm yr^{-1} for contour tillage in soils on 20% slope gradient (Poesen et al., 1997). These rates indicate that downslope tillage would cause 7.2–11.8 cm decrease in surface elevation in 20 yrs, which is the average depth of the entire A horizon in most mountainous soils. Frequent tillage above and below the field boundaries causes dramatic changes in the surface elevation by creating vertical soil banks that can be as high as 4 m, after a few decades (Papendick and Miller, 1977). In Ethiopia, exposed banks at the base of stone bunds were about 0.5 m high after 8 yr of barrier establishment (Nyssen et al., 2000). A systematic cropping of surface rocks on convex areas and gradual migration of rock fragments toward the concavities are evidence of highly eroded sites. Point measurements using erosion pins, paint collars around trees, rocks, and fence posts are also used to measure change in surface elevation by tillage erosion (Hudson, 1995).

5.9.2 Activity of Radionuclides

The spatial distributions/signatures of radionuclides such as ^{137}Cs, ^{210}Pb, $^{239+240}$Pu, and ^{7}Be are used as tracers of tillage erosion (Matisoff et al., 2005; Li et al., 2006). One of the most widely used radionuclides in tillage erosion studies is ^{137}Cs (Walling and Quine, 1991). The ^{137}Cs is the product of wet and dry fallout from nuclear weapon tests that occurred between 1950s and 1970s. The ^{137}Cs fallout from the 1986 Chernobyl accident was mostly restricted to Europe. The use of ^{137}Cs for tracing soil distribution is also common to water erosion studies. The radionuclide ^{210}Pb occurs naturally as a decay product of terrestrial ^{238}U (Matisoff et al., 2002). Beryllium-7 occurs naturally from cosmic rays whereas the $^{239+240}$Pu results primarily from the fallout of nuclear tests. The $^{239+240}$Pu signature is an alternative to ^{137}Cs signature and may even provide more accurate estimates of tillage erosion (Schimmack et al., 2002).

Because deposition/decay of the radionuclides occurs over a long period of time, it is assumed that radionuclides are uniformly distributed within the topsoil. The radionuclides are strongly absorbed by the soil particles and move readily with soil particles during tillage. Thus, the spatial distribution of radionuclides in hillslopes portrays the net effect of soil redistribution. High ^{137}Cs translocation is a signature of high tillage erosion knowing the concentration and redistribution of the ^{137}Cs, which provides estimates of the rates of soil erosion by tillage over a period of time equal to the half life of ^{137}Cs ($\sim$40 yr). Estimation of tillage erosion with the radionuclides is based on the comparison of the spatial distribution of isotope inventories in the tilled soils against that in adjacent untilled soils (e.g, forest or pasture), which for the case of ^{137}Cs is shown in Eq. (5.1) (Van Oost et al., 2005).

$$^{137}Cs_{\text{(residual)}} = \frac{^{137}Cs_{\text{(inventory)}} - {}^{137}Cs_{\text{(reference)}}}{^{137}Cs_{\text{(reference)}}} \tag{5.1}$$

Negative values of $^{137}Cs_{\text{(residual)}}$ indicates soil erosion and positive indicates aggradation or gains. Site-specific calibrations between the distributions of ^{137}Cs inventories and erosion rates must be established in order to obtain quantitative estimates of tillage erosion. The radionuclides are particularly useful to track the historic soil erosion. One of the shortcomings of this approach is that it can not differentiate between erosion by tillage and that by water and wind.

5.10 Measurement of Soil Displacement

Several techniques are used to trace the soil displacement by tillage erosion. One of the common techniques consists in burying tracers prior to tillage operation and recovering them after it. Labeled stone chips, numbered aluminum cubes, and labeled aluminum cylinders are used as soil displacement tracers. The change in position of these tracers as a result of tillage portrays soil shift during tillage, assuming that tracers moved along with soil. Excavation and low-induction electromagnetic (EM) techniques are also used to monitor and measure the displacement of the tracers (De Alba et al., 2006). The EM method is simple, quick, and does not disturb soil like the excavation method. The displacement distance (m) of soil based on the displacement of tracers is computed (Lobb et al., 1995) as per Eq. (5.2)

$$D_d = \int_0^L \left(1 - \frac{C(x)}{C_0}\right) dx \tag{5.2}$$

where L is maximum distance of sampling (m), $C(x)$ is the weight of the tracer (kg) following tillage, and C_0 is the initial weight (kg) of the tracer. The tracer displacement (TD) following a sequence of tillage operations is computed (Van Muysen et al., 2006) as

$$TD = \sum_{i=1}^{n} (a_i - b_i S) T_i \left(\frac{D_i}{D_{\max}}\right) \tag{5.3}$$

where n is number of tillage operations, a and b are regression coefficients between measured soil displacement and slope gradient, S is soil slope, T_i is tillage direction per pass (1 for upslope and 1 for downslope tillage), D_i is tillage depth (m) per pass, and D_{max} is maximal tillage depth (m).

5.11 Tillage Erosion and Crop Production

Changes in soil properties caused by tillage erosion accentuate spatial variability of soil properties and crop yields. Crop yields are generally the lowest in eroded shoulder slopes and the highest in the depositional zones due to significant spatial variations in soil properties (Papiernik et al., 2005). Convex fields often exhibit shallow topsoil layers in contrast with flat terrains. Subsoil horizons exposed by tillage erosion are structurally unstable, high in clay content, and poor in fertility, and are the cause of lower productivity. Poor emergence and delayed establishment of crops on the shoulder slopes are common because of adverse soil structural conditions. Crop yields increase gradually from higher to lower landscape positions because of the greater organic matter content and water retention capacity in concave field positions (Kosmas et al., 2001). The effect of tillage erosion on soil productivity is, however, site specific. Deep soils with thick A horizons are not significantly affected by tillage erosion, but soils with shallow profiles and calcareous horizons and coarse fragments (e.g., gravel, stones), such as those in mountainous areas and dry climates, are easily degraded by tillage erosion, and, in turn, drastically affecting the crop production.

5.12 Management of Tillage Erosion

A number of management strategies are available to control tillage erosion (Table 5.5). Conservation tillage such as no-till systems leaves crop residues on the soil surface and eliminates tillage erosion. Wherever tillage operations are necessary, their prudent management is crucial to reducing tillage erosion. Selection of proper tillage method, performing tillage on the contour direction, and reduction of the number of passes are some of the important management strategies. Tillage methods that minimize the energy spent on tilling soil reduce the magnitude or risks of tillage erosion. Chisel and disk plows cause tillage less erosion than does the moldboard plowing. Implements that invert the soil cause the largest movement of erosion by tillage.

Contour tillage is a conservation effective practice to reduce tillage erosion. It can reduce tillage erosion by 75 and 85% compared to the downslope tillage (Zhang

Table 5.5 Strategies for managing tillage erosion

Tillage operations	Soil slope management
Use:	Use:
• less erosive tillage implements	• reduced tillage and no-till farming
• contour or upslope tillage	• grass barriers on the contour
Reduce:	• alley farming
• tractor speed to 1 $\mathrm{m\,s^{-1}}$	• contour farming
• shallow plowing (<30 cm)	• stone bunds or lines or the contour
• the number of tillage passes	• terraces

et al., 2004b). On steep slopes, the field can be partitioned in smaller segments, and planted to rows of perennial grass barriers or maintained under cover crops to minimize soil transport. Shallow plowing (<20 cm depth) and upslope tillage are also recommended practices to reduce tillage erosion. Practices that minimize tillage erosion to levels equal to or less than caused by upslope tillage must be designed for each sloping field based on experimental data from the same or similar soils. Tillage practices that involve a single tillage pass, disturb minimum depth and use low speed, and use short and flexible implements reduce erosion risks.

In ecosystems where some tillage is essential, planting trees, shrubs, and grasses as contour barriers can reduce soil translocation. Alley cropping and grass barriers are examples of biological soil conservation practices that could reduce translocation by tillage erosion. Contour hedge rows in alley cropping break the slope length in small segments (<20 m) reducing the distance for soil movement by erosion. Similarly, grass barriers/hedges planted perpendicular to the dominant slope can trap sediment and built terraces over time, reducing the field slope and tillage erosion. A study conducted in the Philippines showed that contour ridge tillage and contour farming with grass barriers reduced 30–53% of the total soil erosion (63 Mg ha^{-1} yr^{-1}) in moldboard plowed soils (16–22% slope) (Thapa et al., 1999).

Returning the translocated soil to its original or upslope positions from the deposition zones is one of the approaches to recover soils degraded by tillage erosion. This management approach can be, however, laborious and uneconomical in most cases. The most feasible remediation approach is to minimize tillage, because tillage erosion decreases linearly with decrease in tillage (van Vliet et al., 2001). Methods of tillage erosion control are specific to each soil and climate. These techniques which reduce water and wind erosion may not necessarily reduce tillage erosion. For example, chisel plowing that leaves large amounts of crop residues on the surface to control water and wind erosion can still cause large soil translocation by tillage.

5.13 Tillage Erosion Modeling

5.13.1 Predictive Equations

Numerous predictive equations using slope gradient as an independent variable are available to estimate the magnitude of soil displacement (L) by tillage (Table 5.6). These equations are site specific and thus depend on local soil characteristics and tillage techniques. The simple regression equations for estimating L (cm) are in the form of Eq. (5.4) (Lindstrom et al., 1990, 1992)

$$L = \alpha + \beta(S) \tag{5.4}$$

where S is slope gradient (%), and β and α are regression coefficients. Assuming upslope gradient as positive slope and downslope gradient as negative, the first regression equations developed by Lindstrom et al. (1990) are shown in Eq. (5.5) and (5.6)

Table 5.6 Regression models between soil displacement (L) in m and slope gradient (S) in mm^{-1} (*, ** Significant at the 0.05 and 0.01 probability levels, respectively)

Implement	Tillage direction	Predictive equation	r-square
Moldboard and chisel plow and rotary harrow[1]	Up- and down-slope	L = 6.43– 0.35S	0.72**
Hoe[2]	Downslope	L = 0.54S +0.091	0.83**
Moldboard plow[3]	Up- and down-slope and cross-slope	L = 0.516–0.508 tanS	0.91*
Moldboard plow[4]	Up- and down-slope	L = −0.91S + 0.31	0.72*
	Contour	L = −0.38S + 0.48	
Moldboard plow[5]	Up- and down-slope	L = −0.54S + 0.16	0.73*
	Contour	L = −0.54S + 0.24	
Moldboard plow[6]	Up- and down-slope	L = −22S + 11	0.88**
Ox-drawn ard plow[7]	Contour	L = 0.54S + 0.034	0.84**
Chisel plow (stubble) Chisel plow (pretilled)[8]	Up- and down-slope	L = −0.96S + 0.23; L = −2.18S + 0.41	0.51–0.67**

[1]Van Muysen et al. (2006), [2]Zhang et al. (2004b), [3]Quine and Zhang (2004), [4]Van Muysen et al. (2002), [5]St. Gerontidis et al. (2001), [6]Kosmas et al. (2001), [7]Nyssen et al. (2000), and [8]Van Muysen et al. (2000).

$$L = 34.24 - 1.02(S) \quad r^2 = 0.64 \text{ (parallel to tillage direction)} \tag{5.5}$$

$$L = 44.28 - 1.12(S) \quad r^2 = 0.81 \text{ (perpendicular to tillage direction)} \tag{5.6}$$

Example 1. Estimate the soil displacement by downslope tillage using the simple equations derived by Lindstrom et al. (1990) for a soil with slopes of 5, 10, and 20%.

Displacement in the direction of tillage:

$$L = 63.6 - 2.4(S)$$
$$L = 63.6\text{–}2.4(-5) = 75.6\,\text{cm.}$$
$$L = 63.6\text{–}2.4(-10) = 87.6\,\text{cm.}$$
$$L = 63.6\text{–}2.4(-20) = 111.6\,\text{cm.}$$

Displacement perpendicular to tillage direction:

$$L = 44.28 - 1.12(S)$$
$$L = 44.28\text{–}1.12(-5) = 49.9\,\text{cm.}$$
$$L = 44.28\text{–}1.12(-10) = 55.5\,\text{cm.}$$
$$L = 44.28\text{–}1.12(-20) = 66.7\,\text{cm.}$$

A general model of tillage erosion based on the equation of continuity (Govers et al., 1994) is shown in Eq. (5.7)

$$\rho_b \frac{\alpha h}{\alpha t} = -\frac{\alpha Q_s}{\alpha x} = \frac{\alpha^2 h}{\alpha x^2} \tag{5.7}$$

where ρ_b is bulk density (Mg m^{-3}) of the soil, h is height (m) at a given point of the hillslope, t is time (yr), Q_s is net downslope flux (Mg m^{-1}) due to tillage translocation, and x is distance (m) in horizontal direction. Change in height with respect to change in distance determines the amount of soil erosion. Similar to the splash erosion and soil creep, the net translocation of soil by tillage erosion can be explained by diffusion-type geomorphological attribute. Eq. (5.7) can be solved for Q_s based on L, ρ_b, and tillage depth (D) (m). The Q_s for up- and down-slope tillage direction is estimated (Govers et al., 1994) using Eq. (5.8)

$$Q_s = L \times D \times \rho_b \tag{5.8}$$

Soil translocation or "flux" due to a downslope tillage ($Q_{s,down}$) is

$$Q_{s,down} = L_{down} \times D \times \rho_b \tag{5.9}$$

while soil translocation or "flux" due to a upslope tillage ($Q_{s,up}$) is

$$Q_{s,up} = L_{up} \times D \times \rho_b \tag{5.10}$$

The average net downslope soil translocation per unit width per year, assuming one tillage event per year, is

$$Q_s = \frac{Q_{s,down} + Q_{s,up}}{2} \tag{5.11}$$

or

$$Q_s = \frac{D \times \rho_b \left[(\alpha_{down} + \beta_{down} S) - (\alpha + \beta(-S)) \right]}{2} \tag{5.12}$$

The average soil transport rate or soil "flux" (Q_s) in mass per unit of tillage width along the tillage direction is estimated as

$$Q_s = D \times \rho_b \times (-\beta) \times (-S)_{down} \tag{5.13}$$

or

$$Q_s = kS = -k\frac{\alpha h}{\alpha x} \tag{5.14}$$

where k is the tillage transport coefficient expressed in kg m^{-1}. This coefficient is the most widely used measure of tillage erosion intensity (Govers et al., 1994). The higher the k values, the higher the tillage erosion. The k is specific to each tillage technique, slope gradient, type soil, and tillage depth and speed (Table 5.7). The typical values of k for moldboard plow vary between 200 and 600 kg m^{-1} whereas those for animal- and human-powered tillage implements range from 60 to 120 kg m^{-1}. The k is a critical indicator of tillage erosivity and is an essential input parameter

Table 5.7 Tillage transport coefficient (k) from various selected studies for different soils and tillage implements

Implement	Soil Slope (%)	Depth (m)	k (kg m^{-1})
Moldboard and chisel plow and rotary harrow[1]	0–17	0.05–0.27	167
Moldboard plow[2]	8–12	0.25–0.27	456
Manual hoeing[3]	31–67	0.045–0.06	84–108
Moldboard plow[4]	3–20	0.05–0.10	770
Moldboard plow[5]	15	0.17	265
Moldboard plow[6]	6–22	0.2–0.40	150–670
Chisel plow[7]	30	0.15	225–550
Ard or hoe plow[8]	10–48	0.06–0.11	68
Moldboard plow	3–7	0.23	346
Tandem Disk		0.17	369
Chisel[9]		0.17	275

[1]Van Muysen et al. (2006), [2]Heckrath et al. (2005), [3]Kimaro et al. (2005), [4]da Silva et al. (2004), [5]Quine et al. (2003), [6]St. Gerontidis et al. (2001), [7]Van Muysen et al. (2000), [8]Nyssen et al. (2000), and [9]Lobb et al. (1999).

to compute tillage erosion. The k values have been reported almost in every tillage erosion study.

Assuming that soil translocation by tillage is a diffusion-type process, k is computed (Govers et al., 1994) as

$$k = -D \times \rho_b \times \beta \tag{5.15}$$

$$k = -D \times \rho_b \times (-\beta) \tag{5.16}$$

The Eq. (5.16) indicates that the amount of soil transported increases with an increase in tillage depth, bulk density, and soil displacement.

Example 2. Compute k for a soil that has been recently tilled to a depth of 0.20 and 0.30 m, assuming $\alpha = 0.5$ and $\beta = -2.2$. The bulk density of the tilled soil and slope are 1.45 Mg m^{-3} and 10%, respectively. What would be the magnitude of decrease in k when bulk density decreases to 1.25 Mg m^{-3}.

$$k = -D \times \rho_b \times \beta$$

$\rho_b = 1450\,\mathrm{kg\,m^{-3}}$

$$k = -0.20\,\mathrm{m} \times 1450\,\mathrm{kg\,m^{-3}} \times -1.5\ (\text{m per tillage event}) = 435\,\mathrm{kg\,m^{-1}}.$$
$$k = -0.30\,\mathrm{m} \times 1450\,\mathrm{kg\,m^{-3}} \times -1.5\ (\text{m per tillage event}) = 652.5\,\mathrm{kg\,m^{-1}}.$$

$\rho_b = 1250\,\mathrm{kg\,m^{-3}}$

$$k = -0.20\,\mathrm{m} \times 1250\,\mathrm{kg\,m^{-3}} \times -1.5\ (\text{m per tillage event}) = 375\,\mathrm{kg\,m^{-1}}.$$
$$k = -0.30\,\mathrm{m} \times 1250\,\mathrm{kg\,m^{-3}} \times -1.5\ (\text{m per tillage event}) = 562.5\,\mathrm{kg\,m^{-1}}.$$
$$\text{Magnitude of decrease} = \frac{375}{435} = \frac{562.5}{652.5} = 0.862$$

The k decreases by about 14% as the bulk density of the soil decreases.

Example 3. Calculate rate of soil transport for Example 2 for a soil having a bulk density of 1.45 kg m^{-3}.

$$Q_s = k \times S$$
$$Q_s = 435 \times 0.1 = 43.5\,\text{kg}\,\text{m}^{-1}$$
$$Q_s = 652.5 \times 0.1 = 65.3\,\text{kg}\,\text{m}^{-1}$$

5.13.1.1 Tillage Erosion Rate

The tillage erosion (E) rate (Mg ha^{-1}) can be calculated (Lindstrom et al., 2001) by

$$E = \frac{Q_s}{X} \tag{5.17}$$
$$E = \frac{Q_s, in - Q_s, out}{X} \tag{5.18}$$

where X is slope length (m) under consideration. Net soil loss along a hillslope with various slope segments is predicted using

$$SL = \frac{(-D \times \rho_b \times \beta)\,(S_2 - S_1)}{\Delta x_s} \tag{5.19}$$

where SL is net soil loss from a hillslope, S_1 is slope of first slope segment in adjacent slope segments (m m^{-1}), S_2 is slope of second slope segment (m m^{-1}), and Δx_s is distance between the midpoints of two adjacent slope segments (m).

$$SL = \frac{k\,(S_2 - S_1)}{\Delta x_s} \tag{5.20}$$

Example 4. Calculate the rate of tillage erosion along two adjacent slope segments for the soil in Example 2. The distance between the midpoints of the slope segments is 20 m. The slope of the first and second segment is 20 and 10%, respectively.

$$SL = \frac{k\,(S_2 - S_1)}{\Delta x_s}$$
$$SL = \frac{435\,(0.20 - 0.10)}{20} = 2.175\,\text{kg}\,\text{m}^{-2}$$

5.13.1.2 Deposition Rate

The deposition rate of eroded soils in the concavities is computed (Montgomery et al., 1999) as

$$DR = \frac{A \times \rho_b}{T} \tag{5.21}$$

where *DR* is deposition rate (Mg m^{-1} yr^{-1}), *A* is the cross-sectional area of the depositional zone (m^2), and *T* is the number of years of tillage operations. The ρ_b refers to the bulk density of the translocated soil.

Example 5. The depositional area at the base of a hillslope is 10 m^2 and the density of the eroded soil is 1.30 Mg ha^{-1}. Calculate the amount of soil deposited by tillage erosion over five 5 yr.

$$DR = \frac{A \times \rho_b}{T} = \frac{10 \times 1.30}{5} = 2.6\,\text{Mg}\,\text{m}^{-1}\,\text{yr}^{-1}$$

5.14 Computer Models

While earlier studies on tillage erosion focused primarily on gathering experimental data and computing simple regression models, recent studies are increasingly incorporating sophisticated models accounting for the combined effects of tillage and water erosion. Modeling has enhanced a better assessment soil erosion sources and understanding of tillage erosion processes. Tillage erosion modeling has also benefited from the available data on ^{137}Cs for parameterization of models, and new approaches for simulating tillage erosion across topographically complex soils are being developed.

5.14.1 Tillage Erosion Prediction (TEP) Model

The TEP model is a simple computer model built upon the regression equations of soil translocation by tillage (Lindstrom et al., 2000). It estimates soil translocation for individual hillslope segments and specific tillage operations, assuming a constant slope gradient, uniform soil loss or gain, and no perpendicular soil movement during tillage within each hillslope segment. Soil redistribution simulations for a 50-yr period in Minnesota showed that the TEP model has the capability to identify both eroding and aggrading zones within cultivated hillslopes (Lindstrom et al., 2000). The TEP model in combination with the ^{137}Cs technique is a useful tool to estimate soil degradation by tillage erosion in sloping soils.

5.14.2 Water and Tillage Erosion Model (WaTEM)

The WaTEM simulates the effects of changes in landscape characteristics on water and tillage erosion (Van Oost et al., 2000). The water erosion component is computed using a modified version of the RUSLE while the tillage erosion component is computed using Eq. (5.9) through (5.16). The WaTEM is a topography-driven model and incorporates landscape structure as a major determinant of the tillage erosion. The topographic characteristics are derived from aerial photographs or digital elevation models to perform two-dimensional simulations. Slope gradient, contributing area, surface elevation, tillage depth, tillage transport coefficient, and soil bulk density are essential input parameters. The WaTEM is being used to develop tillage and water erosion rates for various soils and tillage methods and develop tillage erosion maps in combination with ^{137}Cs tillage erosion estimates (Papiernik et al., 2005).

5.14.3 Soil Redistribution by Tillage (SORET)

The SORET is a three-dimensional model that simulates the evolution of soil catena in response to tillage erosion (De Alba, 2003). It simulates the dynamic changes in landscape characteristics, soil profile inversions/truncations and transformations, soil horizon substitutions, surface soil properties, and soil-landscape interactions under tillage erosion. The SORET can predict tillage erosion under different patterns of tillage and model the long-term effects of repeated tillage operations on erosion. The SORET model differs from WaTEM because it performs three-dimensional simulations and computes soil translocations occurring parallel and perpendicular to the direction of tillage under different patterns of tillage (contouring, up and downslope, multiple downslope) in interaction with complex topography using digital terrain models (DTMs). The model estimates soil translocation (d) as per Eq. (5.22) (De Alba, 2003)

$$d = \sqrt{d_{DF}^2 + d_{DP}^2} \tag{5.22}$$

$$d_{DF} = \frac{38.03 - 0.62ST + 0.40SP}{100} \tag{5.23}$$

$$d_{DP} = \frac{41.4 - 0.50SP}{100} \tag{5.24}$$

where d_{DF} is forward soil displacement (m), ST is slope gradient in the direction of tillage (%), SP is slope gradient in the direction perpendicular to tillage (%), and d_{DP} is perpendicular soil displacement (m). Soil redistribution ($Q_{sn(i,j)}$) is computed for a matrix of 3 × 3 grid cells (i, j) in the DTM by Eq. (5.25) through (5.27) based on gains ($G_{s(i,j)}$) and losses ($L_{s(i,j)}$) of soil

$$Q_{sn(i,j)} = G_{s(i,j)} - L_{s(i,j)} \tag{5.25}$$

$$L_{s(i,j)} = \left[\left(d_{DF(i,j)}L\right) + \left(d_{DP(i,j)}L\right) - \left(d_{DF(i,j)}d_{DP(i,j)}\right)\right] D \tag{5.26}$$

$$\begin{aligned} G_{s(i,j)} = {} & \left[d_{DF(i-1,j)}\left(L - d_{DP(i-1,j)}\right)\right] D + \left(d_{DT(i-1,j-1)}d_{DP(i-1,j-1)}\right) D \\ & + \left[d_{DP(i,j-1)}\left(L - d_{DT(i,j-1)}\right)\right] D \end{aligned} \tag{5.27}$$

where is L is length (m) of cell side and D is tillage depth (m). The tillage erosion rate ($T_{e(i,j)}$) per pass in Mg ha^{-1} is computed by Eq. (5.28)

$$T_{e(i,j)} = \left(\frac{Q_{sn(i,j)} \rho_b}{L^2} \right) \times 10000 \tag{5.28}$$

where ρ_b is soil bulk density (Mg m^{-3}).

5.14.4 Soil Erosion by Tillage (SETi)

The SETi model is a process-based approach designed to estimate the tillage transport coefficient as a product of slope gradient, tillage tool, and soil clod displacement interactions (Torri and Borselli, 2002). The SETi considers three phases of clod movement:

1. ***Drag phase***. It refers to the initial stage of soil translocation where, following shearing, the soil material is transported with the tillage implement.
2. ***Jump phase***. It is the stage where soil is ejected by the tillage tool and falls under the influence of gravity and transport velocity.
3. ***Rolling phase***. During this stage, soil clods roll over or slide in response to the gravity until the friction forces overcome the movement where the clods stop.

There are specific set of equations portraying the soil translocation in x, y, and z axis for each phase. Where x is in the direction of the steepest slope along the soil surface, y is transversal to the slope, and z is the vertical axis. The input parameters needed for the model are: mass and diameter of soil clods, the angles of clod trajectory, and speed and direction of tillage.

5.14.5 Water- and Tillage-Induced Soil Redistribution (SPEROS)

The SPEROS process-based model, simulates redistribution of soil as affected by water and tillage erosion based on ^{137}Cs data (Van Oost et al., 2003). SPEROS converts the ^{137}Cs data into rates of water and tillage erosion, and it thus allows the partitioning of relative contributions of water and tillage erosion to total soil erosion. It estimates the soil-profile vertical distribution of ^{137}Cs in the soil profile and lateral translocations of ^{137}Cs by water and tillage erosion. Soil redistribution at locations (a, b) in a grid system is computed using (Van Oost et al., 2003):

$$P(x, y, t) = \int_{-\infty}^{+\infty} \int_{-\infty}^{+\infty} (x, y, t)\, G(a - x, b - y)\, dx dy \tag{5.29}$$

or (Van Oost et al., 2005):

$$E_{till}(k,l) = \rho_b D\left[\left(\sum_{-\infty}^{+\infty}\sum_{-\infty}^{+\infty} G\left(a-x, b-y\right)\right) - 1\right] \tag{5.30}$$

where t is an index for a particular tillage operation, $P(t)$ is two-dimensional ^{137}Cs distribution after t, $S(t)$ is the ^{137}Cs distribution before t, and G_t is two-dimensional tillage displacement probability distribution, D is tillage depth (m), ρ_b is bulk density of the soil (kg m^{-3}). Net soil losses and gains are computed by replacing the ^{137}Cs distribution in Eq. (5.30). The G_t at each point in landscape is calculated using Eq. (5.31) (Van Muysen et al., 2002)

$$d_{long} = \left(g + k_{3,long} S_{long}\right) \frac{TD^{\alpha}}{D_{ref}} \frac{V^{\beta}}{V_{ref}} \tag{5.31}$$

where d_{long} is displacement distance (m) in the tillage direction, *TD* is tillage depth (m), V is tillage speed (ms^{-1}), g, $k_{3,long}$, and α and β are regression coefficients, and D_{ref} and V_{ref} are reference tillage depth and speed, respectively.

5.15 Soil Erosion and Harvesting of Root Crops

An additional but important source of soil erosion and degradation in agricultural systems is harvesting of root crops (e.g., carrots, potatoes, sugar beet, chicory root, leek, cassava, yam, taro, sweet potatoes). Harvesting of root crops does not only increases the soil's susceptibility to erosion by water, wind, and tillage but also causes soil erosion by exporting soil material together with root crops during harvest. Because root crops grow in close contact with the soil, the adhering soil to roots is readily removed along with the products at harvesting. Loose soil and coarse fragments (e.g., stones and gravel) are also removed intermixed with harvested products. Harvest erosion can be as high as those caused by water, wind or tillage erosion, and thus represent another important component of total soil erosion (Table 5.8). Yet, its characterization, importance and implications are largely ignored in soil erosion research.

Table 5.8 Magnitude of harvest erosion

Soil	Crop	Country	Soil Erosion (Mg ha^{-1} harvest^{-1})
Silt loam and loamy sand[1]	Potato	Belgium	0.2–21.4
Across a range of soils[2]	Sugar beet	France	7.7–20.5
		Belgium	4.7–19.4
		Netherlands	3.4–9.8
		Germany	2.2–10.7
Sand, clayey, loamy sand, sandy loam, and silt loam[3]	Sugar beet and chicory roots	Belgium	4.4–19.5
			3.2–12.7
			5–30

[1]Ruysschaert et al. (2006), [2]Ruysschaert et al. (2005), and [3]Poesen et al. (2001).

Similar to tillage erosion, soil erosion by harvest is a human-induced problem and is significant particularly in industrialized farms where introduction of heavy equipment favors cultivation of extensive lands and subsequent increase in soil erosion. Soil erosion by harvest can range between 0.2 and 30 Mg ha^{-1} yr^{-1} (Table 5.8). Studies on the assessment of rates of harvest erosion are confined mostly to Europe. It is estimated that about 6.6 cm of soil has been lost due to root crop harvesting in Belgium during the last 200 yr (Poesen et al., 2001). In contrast with tillage erosion, soil material by harvest is stripped off and is lost permanently, causing a "true" soil loss from the system. The fraction removed is enriched with nutrients and microbial biomass, which cluster around roots by adsorption and symbiosis. Export of essential nutrients affects soil fertility and productivity especially in shallow soils. Losses of total N, available P, and exchangeable K due to sugar beet harvesting can be very high (Oztas et al., 2002). In addition to causing on-site soil degradation problems, soil erosion by harvest can also create off-site environmental pollution (e.g., transport, soil disposal).

Magnitude of soil erosion by harvest depends on the crop type, soil characteristics, harvesting technique, harvesting equipment type used, and climate (Table 5.9). Soil antecedent water content at harvest is the main factor that affects harvest erosion. A study conducted in Belgium showed that soil export by harvest increased linearly with increase in rainfall amount from 140 to 480 mm in the central region (Poesen et al., 2001). Soil texture also influences the magnitude of harvest erosion. The more clayey and wetter the soil is during harvest, the higher is the harvest erosion. Clay and water interact and increase soil adherence to tubers and thus increase in risks of harvest erosion.

Sorting of crops and soil during harvest is critical to reducing soil erosion by harvest. Harvesters with well-designed sorting tables can remove both loose and loosely adhering soil. Improvement in soil tare separation for sugar beets has decreased soil export by harvest. Off-site export of soil by harvesting sugar beets decreased from 6.6 Mg ha^{-1} in 1990 to 3.3 Mg ha^{-1} in 2000 in Germany due to the progress in soil tare separation (Lammers and Stratz, 2003). Harvest erosion can be reduced if the root- or tuber-soil separation is performed on-farm before transporting the harvested products to farmsteads.

Table 5.9 Factors of harvest erosion (After Ruysschaert et al., 2005, 2006)

Soil characteristics	Equipment and operations	Crop characteristics	Climatic factors
• Soil particle size distribution • Water content and drainage • Atterberg limits • Bulk density • Organic matter content • $CaCO_3$ concentrations	• Type and size of harvester • Type of sorting table • Cleaning operations • Harvesting speed • Harvesting depth	• Type of crop • Crop yields • Size and shape of roots • Surface depressions (e.g., potatoes) • Skin roughness	• Rainfall prior to harvesting • Air temperature • Wind

Summary

Tillage erosion is the gradual soil displacement downhill caused by plowing. While importance of water and wind erosion is widely recognized, tillage erosion is also an important component of total soil erosion on sloping croplands. Rates of tillage erosion can be as high as those of water and wind erosion in some soils, and range between 15 and 600 Mg ha^{-1} yr^{-1}. Tillage erosion is a major problem on sloping terrains where agriculture is practiced on soils of 20–80% slope gradients. Similar to water and wind erosion, tillage erosion modifies the soil profile, alters soil properties, and reduces soil productivity. Agricultural modernization with the introduction of aggressive tillage equipment has facilitated the plowing of marginal lands, which has increased risks of tillage erosion.

Rainfall and wind intensity influence water and wind erosion while tillage intensity determines tillage erosion. Tillage operations, unlike water and wind erosion, rarely transport soil off-site, but they redistribute and move soil to the lower landscape positions. Soil slope, soil properties (e.g, antecedent soil water content, soil texture, gravel content and stoniness, bulk density, shear strength), tillage methods (e.g., moldboard plow, chisel plow, disk plow, animal traction, and manual tools), tillage operations (e.g., depth, direction, speed, number of passes) determine the magnitude of tillage erosion. Downslope tillage causes greater erosion than upslope tillage and tillage performed on the contour. Spatial signatures of radionuclides such as ^{137}Cs, ^{210}Pb, $^{239+240}$Pu, and ^{7}Be are used as tracers of tillage erosion. The ^{137}Cs resulting from the fallout from nuclear weapon tests is one of the common radionuclides used in tillage erosion studies. Soil displacement by tillage is also monitored using labeled stone chips, numbered metal tracers, and low-induction electromagnetic techniques.

Reducing the plowing depth and number of tillage passes and establishing conservation tillage, alley cropping, contour farming, and terraces are strategies to reduce tillage erosion. Harvesting of root crops (e.g., potatoes, carrots, sugar beets) is also another source of soil erosion from croplands. As much as 20 Mg ha^{-1} yr^{-1} of soil can be transported off-site with harvested root and tuberous crops. Computer modeling, remote sensing, GIS tools, and statistical modeling are now used for studying soil distribution and magnitude of erosion by tillage.

Study Questions

1. Compute the tillage transport coefficient expressed in kg m^{-1} and rate of soil transport for a soil of ρ_b equal to 1.32 Mg m^{-3} and with 25% slope. The tillage depth was 0.25 m and the soil displacement vs. slope gradient relationship was: $SD = 0.6 - 1.6(S)$.
2. Calculate the rate of soil erosion across various slope segments for the soil in Prob. 1. The average distance between the midpoints of the slope segments is 25 m. The slope of the first, second, third, and four segments is 35, 20, 10, and 5%, respectively.

3. Estimate the tillage deposition rate for a sloping (10%) soil that has been plowed once a year for 5 yr. The bulk density of the soil is 1.40 Mg m^{-3} and the area of the deposition zone is 10 m^2.
4. Compute the soil displacement for all the soils in Table 5.6, assuming a constant soil slope of 15%. Discuss in detail the reasons for the discrepancies in displacement values for the same soil slope.
5. Discuss differences among tillage, water, and wind erosion in relation to factors and processes.
6. Discuss the various types of tillage direction and speed and their influences on soil translocation.
7. How is tillage erosion modeled?
8. Discuss the strategies for tillage erosion management.
9. Discuss the magnitude of soil erosion by crop harvesting.
10. How does the soil organic matter affect the magnitude of harvest erosion.?

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Chapter 6
Biological Measures of Erosion Control

A number of biological and agronomic management practices are available for controlling soil erosion. Important among these are no-till, reduced tillage, crop rotations, cover crops, residue and canopy cover management, vegetative filter strips, riparian buffers, agroforestry, and soil synthetic conditioners. This Chapter discusses the importance of: (1) cover crops (2) crop residues, (3) soil amendments (e.g., manures), and (4) soil conditioners (e.g., polymers) to reducing soil erosion.

There are differences among these biological practices in relation to their mechanisms of erosion control. Biological measures such as crop residues, using manure, and applying conditioners are in direct contact with the soil surface and thus serve as buffers (e.g., residues) or thin films (e.g., conditioners) protecting the soil. In contrast, standing vegetation (e.g., cover crops) reduces soil erosion through the protective effect of its canopy cover which intercepts raindrops above the soil surface and by the mulching effect of residues produced by the growing vegetation.

6.1 Functions of Canopy Cover

Canopy cover is a measure of the fraction of the soil surface covered by vegetation. Different strata of plant leaves and branches make up the multi-storey architecture of plant canopy. Plant canopy acts as a physical barrier against the impacting raindrops. The magnitude of canopy cover determines the proportion of raindrops intercepted by the canopy. Soil erosion is strongly impacted by the canopy cover. In fact, canopy cover is a critical component of the C-factor in the USLE and other soil erosion predictive relationships. Canopy cover reduces soil erosion by intercepting the rainfall and reducing both the kinetic energy of the raindrops and splash detachment. Effectiveness of the canopy cover in controlling soil erosion is affected by the rainfall characteristics, soil properties, and the characteristics of the canopy (e.g., species, height, density).

Plant canopy not only shelters the soil but also interacts with the soil and surface litter underneath to reduce soil erosion. The interaction between plant canopy and surface litter or residues improves the soil hydrological (e.g., water fluxes) and structural properties (e.g., aggregate, stability). Runoff and soil erosion generally decrease exponentially with increase in canopy cover (Bochet and Rubio, 2006). The more the soil is covered with vegetation, the better is the soil protection against erosion. Selection of cropping systems that possess high canopy cover and produce large amounts of surface residue is useful to controlling erosion. Crops with dense canopy grown during periods of high rainfall erosivity reduce erosion risks. Intermittent stands of vegetation with a low canopy cover (e.g., sparse leaves and branches) provide less protection than crops which effectively cover the soil surface. Similar to its effects on water erosion, canopy cover also plays a major role in reducing wind erosion. It intercepts, buffers, and slows the wind velocity. Multistorey canopy enhances resistance against the erosive forces of both raindrops and wind.

6.1.1 Measurement of Canopy Cover

There are several methods of measuring the canopy cover. The simplest method consists of counting the number of centimeters on a meter stick, placed under the canopy at noon, which are shaded or unshaded (Kelley and Krueger, 2005). The percentage of canopy cover is computed by dividing the number of centimeters on the meter stick that were shaded by 100 or the total number of centimeters in a meter. The measurements are normally done along transects during a sunny midday when wind velocity is less than 10 $\mathrm{km\,h^{-1}}$. A similar method of measurement is done with a canopy densiometer, which consists of a mirror with grids to reflect the canopy cover. The grids covered by canopy are counted and divided by the total number of grid points on the mirror to compute the percent canopy cover. There are also optical methods (e.g., high contrast photographs) to measure canopy cover by quantifying the fraction of sunlight that passes through the canopy. Remote sensing methods are new tools for estimating the canopy cover over large areas based on the relationships between canopy cover and spectral vegetation indices and reflectance.

6.1.2 Canopy Cover vs. Soil Erosion Relationships

Canopy cover is an essential input in many models used to predict the soil erosion hazard. Soil erosion and canopy cover relationships are modeled using RUSLE, EPIC, WEPP, SWAT, and other simplified equations. In general, there exists an exponential relationship between soil erosion and canopy covers as is expressed in Eqs. (6.1) and (6.2) (Gyssels et al. (2005)

$$SL = e^{-bC} \tag{6.1}$$

$$RL = e^{-bC} \tag{6.2}$$

where *SL* is relative soil loss, *RL* is relative runoff loss, C is vegetation cover (%), and b is a constant that varies between 0.0235 and 0.0816 for soil loss and between 0.0103 and 0.0843 for runoff loss.

The RUSLE computes the canopy cover influence on soil erosion using Eq. (6.3) (Gyssels et al., 2005)

$$C_c = 1 - f_c e^{-0.1H} \tag{6.3}$$

where C_c is canopy cover subfactor, f_c is fraction of canopy cover, and H is effective fall height. The WEPP model accounts for the canopy cover effect on interrill erosion by multiplying the baseline interrill erodibility (K_i) by a canopy cover subfactor (Zhang et al., 2001).

6.2 Soil Amendments

Soil amendment is defined as any material that is either left on the soil surface or incorporated into the surface layer to decrease runoff and soil erosion while also improving soil properties (SSSA, 2008). Applying amendments on the soil surface is especially effective when used in conjunction with the introduction of conservation tillage systems as opposed to traditional practices where amendments are plowed under. Some soil amendments (e.g., animal manures, crop residues, green manures) have been used since the dawn of agriculture (Table 6.1). They provide innumerable benefits including reduction of soil erosion and improvement in soil physical, chemical, and biological properties.

6.2.1 Classification

Soil amendments can be classified into: organic, natural, and synthetic materials. Natural organic amendments include undecomposed, partly decomposed, and decomposed plant residues. Industrial wastes (e.g., saw dust), municipal wastes (e.g., food wastes), and natural/partly processed materials (e.g., gypsum) are also important amendments. For example, food wastes (>25 million Mg yr^{-1}) account for more than 15% of municipal waste in the USA and can be an important soil amendment when properly composted (Miller, 2002). Composted food wastes stimulate microbial processes, generate and recycle essential plant nutrients, improve soil

Table 6.1 Some commonly used soil amendments

Organic materials	Natural materials/Industrial by-products
• Crop residues	• Paper sludge
• Green manure cover crops	• Sewage biosolids
• Peat	• Lime, dolomite, and flue gas desulfurization products
• Manures	• Gypsum and clays (e.g., vermiculite)
• Sawdust and wood ash	• By-products of biofuel production
• Compost	
• Food waste	

aggregation, reduce soil erosion, and mitigate global warming. Recycling organic by-products reduces disposal costs and constitutes valuable soil amendments.

6.2.2 Specificity

Not all the soil amendments perform the same functions but all contribute to soil erosion control and improvement of soil quality and plant growth. Organic amendments enhance plant growth by improving soil structure, increasing water retention, and replenishing plant nutrients. Some amendments perform specific functions such as lime, which reduces soil acidity or increases pH. Other amendments, such as crop residues, are useful in soil erosion control, nutrient replenishment, soil structural improvement, and soil organic C sequestration.

6.2.3 Soil Conditioner

Soil conditioner is "any material which measurably improves specific soil physical characteristics or physical processes for a given use or as a plant growth medium." (SSSA, 2008). It is a natural or synthetic substance that is added to the soil in small quantities which typically reacts rather rapidly with soil particles to improve one or several soil properties. Polymers are best examples of soil conditioners. Over the last 50 yr, many synthetic water-soluble polymeric materials (e.g., polyacrylamides) have been developed for stabilizing soil and reducing erosion. The new polymers are more affordable, accessible, and effective than the first polymers developed in early 1950's. Polymers are not only useful in reducing soil erosion on croplands but also in stabilizing disturbed urban and road construction sites. Polymers are useful in reducing soil erosion, decreasing non-point-source pollution, improving soil properties, and enhancing plant growth.

6.3 Cover Crops

Cover crops are "close-growing crops that provide soil protection, seeding protection, and soil improvement between periods of normal crop production or between trees in orchards and vines in vineyards" (SSSA, 2008). These are also referred to as green manure crops. The use of cover crops is an ancient practice and dates back to the ancient civilizations in Greece, Rome, China, and others (Magdoff, 1992). Management and role of cover crops have, however, changed over time. In the past, cover crops were either used as animal fodder or plowed under as green manures. Nowadays, cover crops are being promoted as an important companion practice to no-till, reduced tillage, alley cropping, agroforestry, and other conservation practices designed to reduce soil erosion and improve quality of soil and water resources. The new trend is to use cover crop as mulch rather than incorporating it into the soil.

Cover crops are innovative conservation practices, and are specifically grown for:

1. protecting soil against erosion,
2. improving soil properties,
3. enhancing soil fertility,
4. suppressing weeds,
5. fixing N,
6. increasing soil organic matter content,
7. increasing crop yields,
8. recycling nutrients,
9. preventing leaching of nutrients, and
10. improving water quality

Because of multi-faceted benefits, use of cover crops is highly desirable. Cover crops are mainly grown between the cropping seasons. They can also be grown as rotational crops and companions to main crops. Cover crops belonging to graminae or grass species germinate quickly and can trap/catch nutrients from the previous main crops, reducing losses of nutrients by leaching. In addition to scavenging nutrients from the previous crops, cover crops provide essential nutrients to the following crops. For example, legume cover crops supply between 50 and 300 kg ha^{-1} of N, partly if not completely meeting the N requirements of most crops (Sainju et al., 2002). Use of mixed cover crops, including grasses and legumes, increases the biomass return to the soil, enhances activity of soil organisms, and improves soil productivity.

Use of cover crops not only reduces runoff, soil erosion, and use of inorganic fertilizers but also controls weeds, a major constraint in reduced and no-till systems (Fig. 6.1). In temperate regions, winter annuals are the most common cover

Fig. 6.1 Rye as a cover crop for corn-soybean rotation in Pennsylvania (Photo by H. Blanco)

crops. Summer annuals and perennials are also established in some soils. Converting monocropping practices to complex/diverse rotations involving green manure crops is cost-effective and a relatively new paradigm for reducing soil erosion, increasing crop yields, and enhancing soil C sequestration. The use of cover crops has been somewhat constrained by local economic and social conditions, especially in developing countries. If not properly managed, some cover crops can deplete soil water and reduce crop yields (e.g., late- or early-kill). Balancing benefits of cover crops in controlling soil erosion against possible reduction in crop yields is important to assessing short-term economic gains of this conservation-effective measure.

6.3.1 Water Erosion

Establishing cover crops is one of the top conservation practices for reducing runoff and soil erosion from agricultural soils (Table 6.2). Cover crops buffer the erosive energy of raindrops through their dense canopy and stabilize the soil through their roots. This dual function of a cover crop makes it a strategic erosion control practice. Cover crops stabilize and enrich the soil with organic materials. Through biomass input and nutrient trapping, cover crops enhance soil fertility, improve soil structure, and decrease soil erodibility. On steep slopes and in erodible soils, cover crops can reduce soil erosion by as much as one order of magnitude compared to monoculture. Cover crops when used in association with other permanent vegetation (e.g., trees) improve stability and strength of shallow soils and reduce landslides.

Table 6.2 Rates of soil erosion from croplands with and without cover crops

Cover crop	Soil erosion (Mg ha^{-1})	
	Without cover	With cover
Velvet bean[1]	3.3	0.35
Crimson clover[2]		4.42
Rryegrass[2]	11.31	4.08
Lespedeza[2]		5.55
Tall fescue[2]		7.08
Rye and hairy vetch[3]	41.3	3.70
Winter wheat and hairy vetch[4]	74	20
Canada bluegrass[5]		0.42
	2.45	
Downy brome[5]		0.24

[1]Khisa et al. (2002), [2]Malik et al. (2000), [3]Martin and Cassel (1992), [4]Mutchler and Mcdowell (1990), and [5]Zhu et al. (1989)

6.3.2 Wind Erosion

Similar to decreasing water erosion, cover crops also mitigate wind erosion. Cover cropping is useful to control wind erosion in arid and semiarid regions where the soil cover is meager. Growing a cover crop stabilizes soil aggregates and coagulated particles cannot be easily carried by the wind. Cover crops protect soil against wind erosivity between growing seasons when soils are normally denuded and bare. Presence of cover crops increases surface tortuosity and reduces saltation and surface creeping of soil particles during wind erosion. Cover crops can be planted between crop rows perpendicular to the dominant wind direction to provide physical barrier against the wind. A small decrease in wind speed by cover crops results in significant reductions in wind erosion. Wind erosion from soils sheltered with cover crops can be as low as 50% of soils without cover crops (Delgado et al., 1999). Cover crops combined with no-till practices are the most effective means for controlling wind erosion.

6.3.3 Soil Properties

Cover crops also reduce soil compaction and crusting, increase soil macroporosity, and improve soil properties (Table 6.3). The abundant biomass input by cover crops improves soil structure, increases water retention and transmission, facilitates aeration, increases soil fertility, and enhances biological activity. Non-legume and legume winter cover crops are effective at improving soil fertility while providing abundant above- and below-ground biomass to the soil.

Table 6.3 Response of some soil properties to cover crops

Property	Cover crop	Without cover	With cover
[1]Mean weight diameter (mm)	Ryegrass, fall rye, and spring barley	1.2	1.3–2
[2]Hydraulic conductivity (cm h^{-1})	Pigeon pea and mucuna	0.6	1.4–2
[3]Bulk density (Mg m^{-3})	Rye	1.5	1.4
[4]Penetration resistance (MPa)	Carpet grass, creeping grass, guinea grass, elephant grass, style, and Kudzu	0.2	0.1
Macroporosity (%)[4]		17	17–25

[1]Liu et al. (2005), [2]Argenton et al. (2005), [3]Duiker and Curran (2005), and [4]Obi (1999).

6.3.4 Management of Cover Crops

Choice of a cover crop and its management are crucial to harnessing the maximum benefits. Cover crops comprise of annual, biennials, and perennials grasses or legumes. Choice of species and management depend on the specific goals (e.g. erosion control, N build-up, weed suppression). In order to obtain dense stands, cover crops are often seeded at rates higher than grain crops for seed or forage crops for production. Fertilization and use of amendments are also needed in some soils for an optimum growth of cover crops. Incorporation of cover crop as a green manure is recommended prior to blooming. Killing and incorporation of cover crops while foliage is green improve decomposition, increase biological activity, cause a rapid nutrient release, and reduce the C:N ratio of the organic materials. Because cover crops often reduce soil water content, they must be incorporated into the soil several weeks prior to planting the main crops to minimize risks of drought stress in semi-arid and arid regions. Plowing under of cover crops reduces their benefit to soil erosion control as opposed to leaving cover crop mulch on the soil surface that protects the soil against erosion, increases soil organic matter content, enhances nutrient pools, and suppresses weeds.

6.4 Crop Residues

Crop residues are major assets on agricultural soils and provide numerous ecosystem services such as reducing soil erosion, improving soil physical, chemical, and biological properties, increasing crop production, and improving the environment. Specifically, crop residues are critical to reducing runoff and soil erosion, improving soil hydraulic properties, increasing soil water storage, moderating soil temperature, increasing or maintaining the soil organic matter, and improving soil fertility. Crop residues are used for a number of purposes, but their primary function is to conserve soil and water. In some ecosystems, most of the crop residues are used as fodder for animals, while in others residues are left on the soil surface, burned or harvested. Residue management is essential to soil and water conservation, nutrient cycling (e.g., N, P, K, S, micronutrients), and C sequestration. Management practices (e.g., no-till) that leave all or most of the crop residues on the soil surface are preferable.

6.4.1 Quantity

The quantity of crop residue produced varies with cropping system, soil type, and the ecoregion (Fig. 6.2). On global basis, the amount of residue produced for the

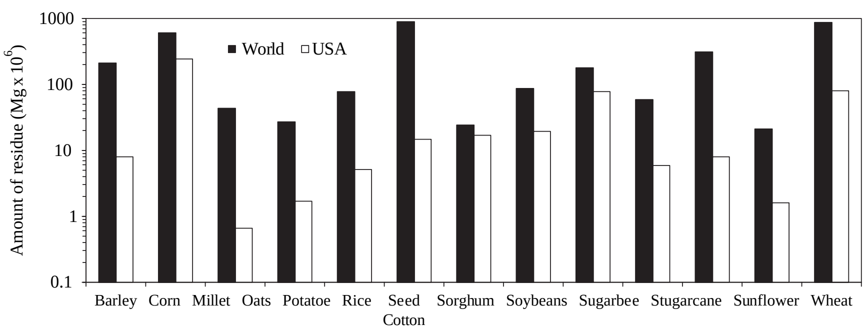

Fig. 6.2 Residue production in the (**A**) world and (**B**) USA for different crops in 2001 (After Lal, 2005)

main crops follows the order: rice>wheat>corn>sugarcane>barley. In the USA, corn residue is the most abundant crop residue and thus most studies on residue management have been focused on corn residue. Global production of crop residues generally increased during the 20th century, but demands of crop residues for competing uses have also increased. The four main competing uses are soil and water conservation, animal feed and bedding, biofuel feedstocks, and industrial raw material.

6.4.2 Soil Properties

Crop residues buffer the soil surface against climatic elements and machinery traffic. They reduce traffic-induced changes in soil mechanical properties such as cone index, shear strength, bulk density, and porosity (Table 6.4). The process of decomposition of crop residues improves: (1) soil's resilience against compactive effects of farm machinery and (2) soil inherent attributes such as biological activity, macroporosity, and water retention properties. Residue cover decreases susceptibility of the surface soil to compression and compaction by reducing surface sealing and crusting, decreasing rainfall-induced consolidation, and decreasing susceptibility to abrupt wetting and drying (Fig. 6.3). While soil bulk density decreases, water retention and aggregate stability increase with application of crop residues. Improved macroporosity under residue cover increases the saturated hydraulic conductivity and water infiltration capacity.

Hydraulic conductivities in no-till soils with complete residue cover can increase ten-fold compared to soils without residue cover (Blanco-Canqui et al., 2007). The most significant effect of residue management is on the energy balance dynamics. Residue cover reduces the abrupt fluctuations of soil water and temperature regimes. No-till soils with residues often have higher water reserves than those without crop residues. Temperature of no-till soils with residue mulch can be lower in spring and summer compared to soils without crop residue mulch. Evaporation in no-till soils decreases with increase in the rate of residue retention, thus increasing plant-available soil water reserves.

Table 6.4 Influence of crop residues on near-surface physical properties of a silt loam [After Blanco-Canqui et al., (2006) and Blanco-Canqui and Lal (2007)]

Property	Without residues	With residues
Bulk density (Mg m^{-3})	1.2a	1.1b
Cone index (MPa)	1.2a	0.9a
Soil porosity (mm mm^{-1})	0.5b	0.6a
Mean weight diameter (mm)	1.5b	2.6a
Tensile strength of aggregates (kPa)	56b	252a
Saturated hydraulic conductivity (mm h^{-1})	0.3b	3.2a
Plant available water content (cm)	0.7b	1.5a
Air permeability (μm^2)	0.1b	27a
No. earthworm middens (per m^{-2})	0.0b	160a
Soil organic matter content (g kg^{-1})	33b	49a

Fig. 6.3 Crop residues protect soil from cracking, crusting, and surface sealing (Photo by H. Blanco)

Residue management can greatly impact soil's dynamic properties, but the magnitude of change depends on soil type, residue amount, tillage systems, and climate. Changes in residue cover may have higher effects on properties of silt loam than those of clayey soils because of differences in drainage and residue decomposition rates. Tillage and climate affect residue decomposition and the amount of soil organic matter accumulation, which, in turn, impacts soil physical, chemical, and biological properties.

6.4.3 Runoff and Soil Erosion

Losses of runoff and soil organic matter -enriched sediments from unprotected cultivated soils on steep terrains can be high. Leaving crop residue on the soil surface significantly reduces runoff and soil erosion (Table 6.5). Complete removal of residue results in rapid initiation runoff and higher runoff velocity. Total runoff and soil erosion from plowed soils without residues are several orders of magnitude higher

Table 6.5 Selected studies showing the impacts of crop residues on water erosion

Residue type	Residue (Mg ha^{-1})	Soil erosion (Mg ha^{-1})	
		Without residues	With residues
Hay[1]	2.25	5.6	0.8
	4.50	5.6	0.4
	9.00	5.6	0.1
Corn[2]	5.6	17	10
Wheat[2]	10.4	17	1.7

[1]Rees et al. (2002) and [2]Mcgregor et al. (1990).

than those from no-till soils with residues regardless of the soil type. Runoff and soil erosion from residue mulched soils are the lowest of all cultivated soils. Reduction of runoff in soils with crop residue is because of the high water infiltration rate and macroporosity. Reduction of water runoff and soil erosion in mulched soils also reduces off-site transport of non-point source pollutants (e.g., fertilizers, pesticides, and herbicides) to rivers and streams. Presence of crop residues is more effective in reducing soil erosion and sediment-bound chemicals than in reducing water runoff. Maintaining residue cover significantly reduces losses of plant nutrient (NO_3–N, NH_4–N, and PO_4–P) losses in runoff. Nutrient concentration in runoff water decreases linearly with increase in the amount of crop residue mulch. Crop residues used in conjunction with conservation tillage systems (e.g., no-till) are highly effective practices in reducing soil erosion from agricultural soils.

6.4.4 Crop Production

Crop residue mulch controls the primary factors affecting plant growth including soil water and temperature regimes, light or net radiation, biological activity, and their interactive processes particularly near the soil surface. Decomposing crop residue materials improve soil structure and fluxes of water, air, and nutrients in the soil. Crop yields increase linearly with increase in the rate of residue return due to increased nutrient input and improvements in soil structure and related properties. Differences in rate of crop residue applications explains $>$ 80% variability in crop yield mainly because of differences in soil water and soil temperature regimes.

It is important to note that while residue retention is essential to reducing soil erosion, mulching may not always improve crop yields. Residue mulch may increase, have no effect, or even decrease crop yields, depending on soil type, tillage management, and the prevailing climate. Residue mulch is particularly essential to plant growth in dry years or arid climates when it reduces soil evaporation and conserves water. Excessively wet and cold conditions during the seedling stage, inadequate control of weeds and pests, low pH and nutrient deficiency with high rates of increased crop residue mulch can reduce crop yields (Mann et al., 2002). Low soil temperatures beneath a dense residue cover may delay planting while decreasing and slowing seed germination. There is an optimum range of soil temperature for every crop. Planting in mulched soils must be done when soil temperature at seeding depth reaches or exceeds the required minimum temperature. Because surface soil warms up more rapidly, shallow seeding may be a strategy for increasing seed germination. Soil temperature controls many physical, chemical, and biological reactions essential to germination. Biological decomposition of organic compounds and fluxes of water, air, and heat are slow when temperatures are sub-optimal during germination. Supra-optimal soil temperatures can also adversely affect processes of germination by reducing biological activities and nutrient uptake.

Residue mulch may also provide habitat for rodents, insects, and pathogens. Shredding residues and use of crop rotations are recommended practices to counteract problems associated with dense residue cover. Proper crop residue management can, however, increase crop biomass and grain yields by reducing temperature fluctuations, improving nutrient and water availability, and enhancing the soil fertility required for root growth and proliferation. High rates of residue retention can delay seedling emergence and reduce plant height during the early period of growth, but, later in the season, plant heights between mulched and un-mulched soils even out and may reverse because of favorable soil water and temperature regimes in mulched soil. Residue removal can adversely affect grain and biomass yields on sloping and erosion-prone soils more than on clayey soils on gentle slopes.

6.5 Residue Harvesting for Biofuel Production

Concerns over increase in the fuel costs and global warming caused by the atmospheric CO_2 abundance are among important factors underpinning energy entrepreneurs to develop alternative and renewable fuel. Production of cellulosic ethanol based on renewable biomass or crop residues is one such option. For example, in the USA about 1.3 billion dry tons of crop residues grown annually can produce 130 billion gallons of ethanol assuming that 100 gallons of ethanol can be produced per ton of corn residues (Perlack et al., 2005). Residues of cereal crop (e.g., rice, wheat, corn, millet) are potential lignocellulosic biomass feedstocks for ethanol production (Fig. 6.4). Total amount of crop residues produced in the world

Fig. 6.4 Corn produces large amounts of residues (Photo by H. Blanco)

is estimated to be about 4 Pg (1 Pg = petagram = 10^{15} g = 1 billion metric ton = 1 gigaton), and one gigaton (GT) of residue can produce 0.25–0.30 gigaliter (GL) of ethanol (Lal, 2006). Attention is particularly being focused on corn residues as a preferred feedstock because of its high cellulose (∼70%) and lignin (∼20%) contents, when compared with other crop residues (Wilhelm et al. 2004). Several ethanol plants are envisaged and soil building crops such as legumes and other perennials are being replaced by corn as price of corn and cost of fuel increase. Energy entrepreneurs are planning to harvest corn residue, and significant advances are being made in fermentation processes of corn residue using enzymes to produce ethanol from cellulose.

While production of liquid biofuels from biomass is a plausible goal to reduce the excessive dependence on fossil fuels and decrease the net emissions of greenhouse gases, indiscriminate removal of crop residue for biofuel production, however, reduces the amount of biomass left on the soil, and may have detrimental effects on soil conservation and agronomic productivity. Retention of crop residue is important to soil erosion control and sustained crop production (Lal, 2006).

Removal of residues can:

- deteriorate soil properties,
- reduce soil organic matter concentration,
- increase emissions of greenhouse gases,
- alter soil water, air, and heat fluxes,
- reduce grain and biomass yield,
- accelerate soil erosion,
- reduce microbial activity,
- deplete plant nutrients, and
- increase risks of non-point source pollution.

6.5.1 Threshold Level of Residue Removal

In some soils and ecosystems, it might be possible to remove a portion of crop residues for energy production and other purposes without adversely affecting soil functions. Information is lacking on the maximum permissible removal rates of residues while maintaining desired level of soil productivity, crop production, and environmental protection. Data from some experiments indicate that about 30 and 40% of the total corn residue production in the U.S. may be available for biofuel production (Graham et al., 2007). However, these estimates are based only on the residue requirements to reduce soil erosion risks, and not based on the needs to enhance productivity and increase soil C sequestration. The maximum amount of crop residue that can be removed in the U.S. Corn Belt region must be based on soil erosion risks, need for C sequestration, and the necessity to reduce non-point source pollution and minimize the dead zone in the Gulf of Mexico and other coastal ecosystems.

The impacts of crop residue removal on soil properties, crop yield, soil erosion and water runoff under different tillage systems are soil specific. Thus, the fraction of crop residue available for removal is indeed site specific. Maximum collection rates of crop residue must be determined by soil type and ecoregion prior to undertaking large scale crop residue harvesting for ethanol production. Specific recommendation guidelines on residue removal rates must be developed under site-specific and contrasting soil types, tillage methods, and ecosystem characteristics.

6.5.2 Rapid Impacts of Residue Removal

Changes in soil properties as a result of residue removal can be rapid, depending on the soil and ecosystem. A study conducted on the residue management in Ohio showed that changes in near-surface soil physical properties (e.g., crusting, soil strength, and water content) were immediate when 25, 50, 75, and 100% of residue cover from no-till continuous corn was removed from three contrasting but representative soils in northeastern, northwestern, and western Ohio (Fig. 6.5). The data from these sites showed that excessive or complete residue removal reduces soil porosity, exacerbates surface crusting and sealing, increases soil compaction, and reduces soil organic matter content even within one-year since initiation of residue removal. Crop residue removal for biofuel production is not a sustainable practice in most soils (Blanco-Canqui and Lal, 2007).

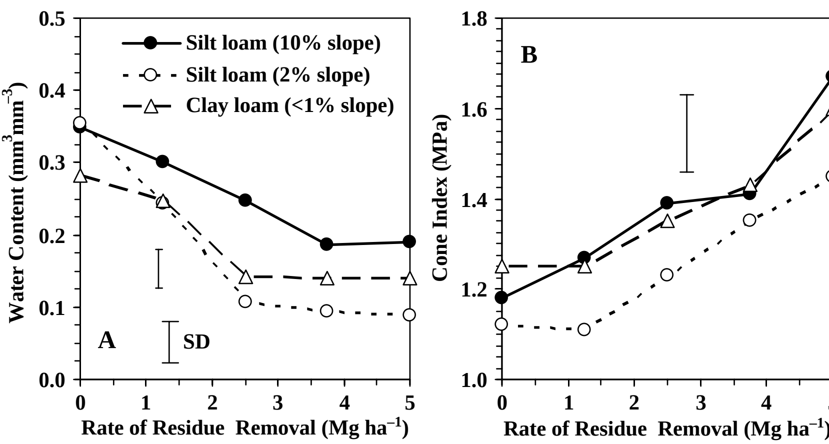

Fig. 6.5 (**A**) Soil water content decreases and (**B**) soil compaction increases with increase in corn residue harvesting across three contrasting soils in Ohio (After Blanco-Canqui et al., 2006)

6.6 Bioenergy Plantations as an Alternative to Crop Residue Removal

Because excessive removal of crop residues exacerbates soil erosion and adversely affects soil properties, biomass feedstock for biofuel production must be produced from dedicated or specific energy plantations established on non-prime agricultural soils (e.g., surplus land, marginal croplands, degraded soils, minesoils, wastelands). Establishing bioenergy plantations is a viable alternative to removing crop residues from agricultural soils. Short rotation woody perennials such as willow and poplar and perennial warm season grasses such as switchgrass, Indian grass, and big bluestem are suitable for establishing bioenergy plantations because of their high biomass yield, rapid growth, low-maintenance, perenniality, and high adaptability to diverse soils and ecoregions (Sanderson et al., 1996). These high-biomass yielding crops can eliminate the possible need of replacing food crops with high cellulosic plantations. The warm season grasses are persistent, and are particularly suitable for adaptation to marginal soils and ecosystems where soil stabilization, erosion control, are needed. Most warm season grasses have extensive deep (>1.5 m) root systems.

The land area needed for establishing energy plantations may compete with that needed to grow food crops. Thus, establishing energy plantations on agriculturally marginal soils could be beneficial to reducing the competition for land. Most importantly, growing warm season grasses as bioenergy crops may be particularly important in soils and ecoregions where stover removal adversely impacts soil characteristics. Information on the performance of warm season grasses on agriculturally marginal soils and reclaimed minesoils is critical to growing warm season grasses as biofuel feedstocks to produce ethanol. Restoration of degraded soils, marginal croplands, and mined soils by establishing bioenergy plantations is also an important strategy for producing bioenergy feedstock while reducing soil erosion, improving soil properties, and mitigating climate change.

The principal task is to further assess the potential sources of renewable biomass for biofuel production based on experimental data. The increased impetus to replace the dependence on fossil fuels by 25% with biofuels within the next 20 yr creates an opportunity to develop advanced bioenergy crops and improve biorefining technologies for conversion of cellulosic biomass to transportation biofuels (US-DOE, 2006). Developing renewable energy alternatives requires a coherent and integrated mission among energy industries, biomass producers, and biotechnological industries.

6.7 Manuring

Use of manure is one of the ancient practices to improve crop production and enhance soil fertility (Fig. 6.6). Manure is very rich in organic matter and macro- and micro-nutrients essential for plant growth. Both solid and liquid animal manures are used as fertilizers. Manure is either knifed into the soil or spread on the soil surface prior to sowing crops. Dried manure of animals from corral or manure mounds has been used for centuries to fertilize soil long before the inorganic or commercial fertilizers were developed. Manure from sheep, cattle, and poultry is among the common types of animal manure. Manuring not only improves crop production but also improves soil properties and reduces soil erosion.

Fig. 6.6 Spraying animal manure slurry is common for improving the soil fertility (Courtesy USDA-NRCS). Manure application at optimum rates is an important to reducing risks of water pollution

6.7.1 Manuring and Soil Erosion

Manuring reduces soil erosion by increasing formation, stability, and strength of aggregates due to the addition of organic matter. Organic matter-enriched aggregates are less susceptible to slaking and have higher inter- and intra-aggregate macroporosity, which results in higher water infiltration rates. Manuring can reduce water runoff by 70–90% and sediment loss by 80–95% as a result of increased organic matter content (Grande et al., 2005). Using manure in combination with other conservation practices, such as no-till with high retention rate of crop residues, is an effective strategy for reducing soil erosion.

Indiscriminate use of manure may have detrimental impacts on water quality. Thus, optimization of the rate of manure applications is important to reducing soil erosion and minimizing pollution. In well-drained soils, manure applications can reduce nutrient losses in water runoff by increasing infiltration rate and improving soil structure. The transport of soluble nutrients from manured no-till soils is often lower than from manured tilled soils. Omission of tillage interacts with manuring and surface residue mulch in reducing nutrient losses in water runoff. Establishing grass barriers on sloping croplands is also a useful recommended measure to minimize off-site transport of manure-derived pollutants.

6.7.2 Manuring and Soil Properties

Manuring decreases soil compaction and increases soil self-mulching capacity. It modifies the soil matrix by buffering the excessive consolidation of soil dry aggregates and by improving the overall structural strength of the soil. Combination of manuring with no-till farming improves soil properties more than plowed systems with manure. Manuring not only improves soil properties at the macro-scale but also at the microscale. Manuring decreases bulk density, cone index, and shear strength of the soil (Table 6.6). Aggregates of manured soils have lower tensile strength and higher water retention capacity compared with unmanured soils. The higher water adsorption capacity increases the plant available water reserves. Manure additions reduce soil strength by improving soil structure, enhancing biological activity, and promoting aggregation and formation of macropores. Manure has elastic properties and buffer soil against compaction and densification. Manure application activates a range of microbial processes essential to soil function. It enhances bioturbation by earthworms and other fauna, reduces soil compaction, and increases soil resistance to raindrop and runoff erosivity. When managed properly, animal manure reduces demands for fertilizers and improves crop productivity.

Table 6.6 Manuring impacts on soil properties on a 35-yr no-till management on a sloping and erosion prone soil [After Blanco-Canqui et al. (2005) and Shukla et al. (2003)]. Values accompanied with the same letter within each row are not significantly different

Property	Without manure	With manure
Bulk density (Mg m^{-3})	1.21a	1.09b
Cone index (MPa)	0.64a	0.35b
Soil porosity (mm^3 mm^{-3})	0.54b	0.59a
Mean weight diameter (mm)	2.14b	3.76a
Saturated hydraulic conductivity (mm h^{-1})	0.08b	0.37ab
Cumulative infiltration (cm)	86.7ab	104.1a
Water content at 0.3 bar (kg kg^{-1})	0.26b	0.35a
Soil organic matter content (g kg^{-1})	30.50b	86.03a

6.8 Soil Conditioners: Polymers

A polymer is a natural or synthetic compound of usually high molecular weight that consists of various millions of inter-connected monomers or long chains of molecules (Martin, 1953). Polymers are commonly known as plastics or resins produced from natural gas. The potential of using polymers in agricultural soils to improve quality of surface soil is high. Interest in the use of polymers started first in the USA in early 1950s (Allison, 1952). Vinyl acetate maleic acid (VAMA) known as Krilium or CRD 186, hydrolyzed polyacrylonitrile (HPAN) or CRD 189, and isobutylene maleic acid (IBM) were some of the first water-soluble polymers used as soil conditioners in the 1950s and 1960s (Nelson, 1998). Krilium was the most broadly advertised polymer under the labels "Friendly Soil" and "Year-Round Soil Conditioner" (Martin, 1953).

The introduction of these polymers in early 1950's created an unprecedented interest in what seemed to be a chemical solution to all soil degradation problems such as compaction, crusting, surface sealing, water runoff, and accelerated soil erosion. Nevertheless, the high cost, difficulties in use, expensive methods of applications, and mixed field results led to disappointments, resulting in the eventual abandonment of these polymers. Subsequent research in the following years has considerably benefited from the early works on polymers and focused on the development of more user-friendly polymers. Polymers including Bitumen and Sarea were introduced in late 1970s and early 1980s and became relatively popular particularly in slope stabilization along roads and highways (Wallace and Wallace, 1986).

Two widely used bitumen emulsions to stabilize soil, reduce soil erosion, and improve plant growth are anionic and cationic forms. Anionic bituminous emulsion ”Bituplant” combined with ”Sarea Evaporation Inhibitor” reduces soil evaporation, loosens compacted soils, promotes aggregation, and improves soil water retention, germination, root growth, and crop yields. Cationic polysaccharides (PSDs), resulting from transformation of organic matter, are also conditioners used for soil stabilization and erosion control (Graber et al., 2006).

6.9 Polyacrylamides (PAMs)

Polyacrylamides (PAMs), polymers with high molecular weights, are used to reduce soil erosion particularly in irrigated soils (Wallace and Wallace, 1986) (Fig. 6.7). PAMs have a wide range of molecular weights and formulation types and can be cationic, anionic, and nonionic. Anionic PAMs, water-soluble compounds with about 150,000 monomers per molecule, are used for erosion and runoff control (Sojka et al., 2004). More than 400,000 ha of irrigated soils in the USA are treated with PAMs, and the largest treated area is in Idaho (Sojka, 2006). The development of PAMs with high molecular weight has reduced costs of purchase and rates of application and improved the methods of application. The high application rates (500–1000 kg ha^{-1}) of PAM used in early studies have been reduced to 10–20 kg ha^{-1} while still achieving the same results of soil erosion control. Reduction in the rate of application of PAM is attributed to the advancement in chemistry of synthetic polymers (Terry and Nelson, 1986).

Compared with Krilium, PAM is a better soil conditioner because the amount of PAM needed to achieve the same or even better results of soil protection is 10–100 times lower. In the 1950’s, polymers were commonly plowed under to a depth of 10 or 20 cm. Presently, PAM is typically applied on the soil surface and is not incorporated into the entire plow layer. Surface application lowers the application rates, decreases the costs, and makes PAM economically more attractive to land managers. PAM forms thin, porous films on the soil surface, acts as a blanket to protect soil from the soil erosive forces. Anionic PAM is an environmentally safe polymer and does not pose a threat to either soil organisms or aquatic life (Sojka, 2006) Use of PAM technology is increasing particularly in regions with furrow and sprinkler irrigated soils. In the USA, scientists at the USDA- Northwest Irrigation and Soils Research Laboratory (NWSRL) and USDA-National Soil Erosion Research Laboratory (NSEL) began researching on PAM during early 1990’s and have generated ample information on PAM performance for controlling runoff and soil erosion from irrigated croplands and construction sites. Polyacrylamides

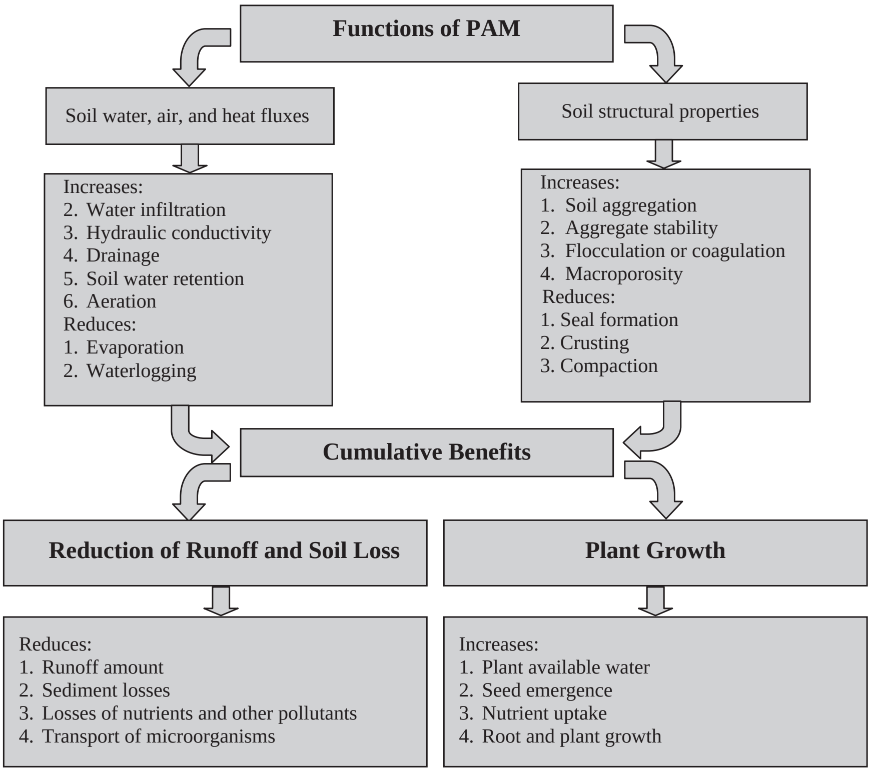

Fig. 6.7 Benefits of PAM used for soil and water conservation on agricultural soils

are also important to coagulate and remove nutrients, pesticides, microorganisms, and weed seeds from water runoff (Sojka, 2006). PAM has many expanded uses. Aside from reducing erosion control, PAM can improve drainage, enhance removal of salt, sediment, and NPS source pollutant (Sojka, 2006), and increase plant available water for seed emergence and crop establishment in coarse textured soils across semi-arid and arid soils where water is extremely scarce for crop production (Sivapalan, 2006). PAM is also beneficial to flocculate suspended sediment and reduce turbidity in stormwater from urban areas. Blanco-Canqui et al., (2004) showed that application of PAM at a rate of 9 kg ha^{-1} significantly reduced runoff and soil erosion (Fig. 6.8).

6.9.1 Mechanisms of Soil Erosion Reduction by Polyacrylamides

Polyacrylamides reduce soil erosion by:

- stabilizing soil aggregates,
- dissipating the kinetic energy of rain,

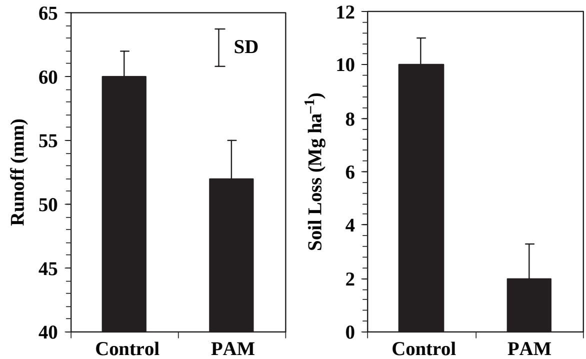

Fig. 6.8 Application of PAM reduced runoff and soil loss on a silt loam (After Blanco-Canqui et al., 2004)

- maintaining the soil surface roughness,
- interacting with inter-aggregate spaces,
- increasing the cohesiveness of soil particles,
- decreasing soil detachment,
- reducing surface sealing and crusting,
- flocculating suspended soil particles,
- stabilizing water conducting macropores,
- reducing dispersion of clay particles, and
- forming bridges of inter-particles

Principal mechanisms of soil stabilization by PAM are:

- adsorption of PAM molecules by clay edge surfaces
- flocculation of soil particles through the reduction of electrostatic repulsion forces among the adjoining particles.
- Interaction of PAM with clay particles and formation of chemical bridges and aggregates.

Reduction in aggregate breakdown decreases the amount of non-flocculated soil particles available for clogging soil pores and erosion. These interrelated processes improve soil hydraulic properties, reduce runoff, increase infiltration rate and hydraulic conductivities, and improve plant growth and crop yields (Fig. 6.7). The PAM molecules do not penetrate into the soil aggregates but remain mostly on the surface. Thus, PAM does not alter the internal soil structure. It improves only the surface structural characteristics, which increases infiltration and reduces runoff. PAM-treated soils resist raindrop impacts and detachment due to increased aggregate stability. Surface applications of polymers improve crop emergence by reducing slaking, crusting, and increasing stability of aggregates. The PAM additions stabilize the existing soil structure and enhance pore continuity and abundance but do not improve soil structure unlike organic amendments (e.g. green manures, crop residues). Application of PAM to compact or degraded soils may improve water movement within the upper few centimeters. PAM may not significantly improve cohesion and stability of coarse textured soils but can reduce excessive water infiltration and increase water retention capacity.

6.9.2 Factors Affecting Performance of Polyacrylamides

Effectiveness of PAM for reducing soil erosion depends on a number of interactive factors including soil type, PAM properties, and rainfall and runoff characteristics and soil management (Table 6.7).

Table 6.7 Factors affecting the performance of PAM

Soil characteristics	Polyacrylamide characteristics	Rainfall/irrigation patterns	Soil management
• Slope and texture • Clay mineralogy • pH and ionic strength • Types of soil ions • Surface conditions • Organic matter content • Salinity and sodicity	• Molecular weight • Charge density • Composition • Type (e.g., emulsion)	• Intensity and amount • Types of irrigation • Frequency of rains and irrigations • Quality of irrigation water	• Tillage methods • Residue cover • Grass strips • Use of other amendments

6.9.3 Soil Characteristics

Soil texture is one of the main factors that affect PAM performance. Water-soluble PAM performs the best on fine-textured soils because PAM molecules readily interact with soil colloids and fine particles to form floccules. PAM molecules are attracted by coulombic and Van der Waals forces to the surface of fine particles, which have higher specific surface area. The enhanced attractions improve particle cohesion and resistance to shearing forces by runoff. Clay minerals exert a significant effect on PAM sorption which is in the order of montmorillonite > kaolinite > fine sand in accord with the specific surface of soil materials. Presence of ions at differing concentrations can alter the PAM sorption ability of clay minerals. Soils with abundant divalent cations such as Ca^{2+} and Mg^{+2} are more effective in PAM sorption than soils with monovalent cations such as Na^{+}. Size, internal structure, and electrostatic charge of clay minerals explain differences in PAM sorption by soil surface. Salt content of the soil solution or irrigation water is an important factor affecting PAM performance because increase in salt content decreases the amount of water adsorbed by PAM molecules. Organic matter reduces the PAM adsorption rates significantly because of the reduction of sorption sites and increase of electrostatic repulsion between the soil particles.

6.9.4 Polyacrylamide Characteristics

There are a variety of PAM formulations with different molecular weights, ionic charges, and forms which determine the PAM effectiveness. PAM formulations include dry granular beads, blocks, powders, and liquid or emulsion. The negative charge density of PAM varies between 2 and 30% with a typical value of 18% (Sojka, 2006). The dry forms have about 80% of active ingredient by weight while the emulsions have 30 or 50% (Holliman et al., 2005). The PAM used for infiltration improvement often has low molecular weights. The soil stabilization is a function of molecular weight and degree of hydrolysis of PAM. The higher the molecular weight and the lower the degree of hydrolysis, the greater the soil aggregate stabilization. Sprayed PAM may control the soil erosion better than dry applied PAM in the early stages following the onset of rains because of rapid interaction of emulsions with soil (Peterson et al., 2002).

The two common forms of PAM include: (1) water soluble and (2) non-water soluble or cross-linked PAMs (Holliman et al., 2005). The water-soluble PAMs are also called “linear” and “non-crosslinked” and are commonly used for erosion control. Although cross-linked or non-linear PAMs are insoluble in water, it can adsorb significant amounts of water, a property that makes them likely amendments for improving the water retention capacity of sandy soils. The development of cross-linked polymers has increased use of polymers for increasing water retention in coarse-textured soils. Cross-linked polymers can absorb water 100–1000 times their dry weight. The kinetics of water holding capacity can be estimated using Eq. (6.4)

$$C_w = \frac{C_{w,\max}\left(\dfrac{t}{T}\right)^n}{1+\left(\dfrac{t}{T}\right)^n} \tag{6.4}$$

where C_w is water capacity of the polymer at 20°C (g g^{-1}), $C_{w,max}$ is water capacity of the polymer at swelling equilibrium sate, t is time (min), T is time necessary to obtain 50% swelling, and n is a constant based on temperature and structure of the material (Bouranis, 1998). A high rate of PAM application does not necessarily increase its effectiveness. Initial applications of PAM may have greater effect on reducing soil erosion than subsequent heavy applications.

6.9.5 Rainfall/Irrigation Patterns

Effectiveness of PAM is also a function of rainfall intensity and irrigation patterns. The higher the rainfall intensities, the shorter the longevity of PAM for soil erosion control. Because PAM effectiveness diminishes with time, greater amounts of PAM or split applications may be needed to reduce soil erosion and water runoff over one or various seasons. The PAM effectiveness for reducing soil erosion can decrease even within a short time after application under intense rain storms. Beneficial effects of PAM application at 2–4 kg ha^{-1} may only last for one or two irrigation/rain events. Effectiveness of PAM can decrease even within one hour following PAM application depending on the rainfall intensity and PAM amount (Fig. 6.9).

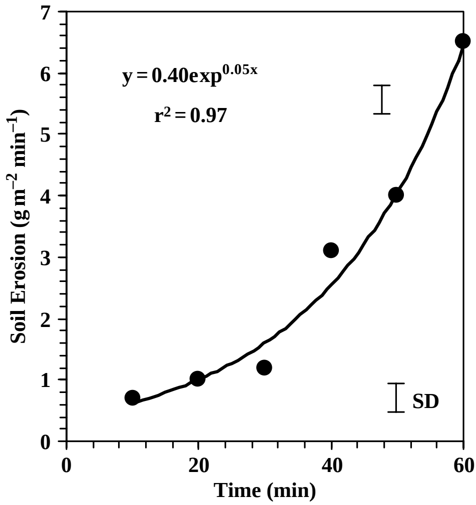

Fig. 6.9 Soil erosion from berms treated with 9 kg ha^{-1} of PAM under 69 mm h^{-1} of simulated rainfall (After Blanco-Canqui et al., 2004). The error bar is the standard error of the mean

6.9.6 Soil Management

Use of PAM in combination with other soil erosion control practices improves performance of PAM for controlling soil erosion from disturbed sites. Common practices include using PAM in combination with: (1) gypsum, (2) crop residues, and (3) grass buffer strips. Applying PAM in conjunction with other practices also makes the use of PAM more adaptable to diverse soil types and climatic conditions. For example, applying crop residue mulch to PAM treated soils can double the reductions in soil erosion compared to PAM alone (Bjorneberg et al., 2000). Combination of PAM with other practices is particularly important to improving PAM performance in highly disturbed sites with steep slopes. PAM applications at low rates may not be very effective at reducing turbidity and sediment losses from steep slopes at construction sites, but addition of mulch and establishment of grass can improve PAM performance. It is important to note that PAM is not a substitute for other conservation practices. Polymers are best suited for temporary stabilization of freshly tilled or disturbed soils while vegetation or other permanent conservation measures are becoming established.

6.9.7 Polyacrylamide vs. Soil Water Dynamics

PAM can either increase or decrease water infiltration depending on the soil. On soils dominated by clay or silt, application of PAM commonly increases water infiltration rate, thereby reducing runoff and soil erosion. The improvement of water infiltration in fine-textured soils by PAM is caused by the increased flocculation, decreased aggregate detachment and clogging of pores, and increased surface-connected macropores. On sandy soils, in contrast, PAM slows water infiltration and improves soil water retention. The viscosity of water increases rapidly with additions of PAM, which causes reduction in water infiltration. Reduction of infiltration in sandy soils means less irrigation and thus reduction in irrigation costs. The PAM-induced increases in soil-water retention capacity in sandy soils can be beneficial to crop growth through increase in the amount of water available because the low water retention capacity and excessive deep percolation reduce the efficiency of water and fertilizer use by plants in coarse-textured soils. The cross-linked PAMs swell up to 100–1000 times their dry weight by absorbing water (Sivapalan, 2006). One g of cross-linked PAM can absorb 10–1000 mL of water depending on the PAM and soil characteristics. Soil water retention capacity by cross-linked PAMs increases between 20 and 50% with increase in PAM additions in sandy soils.

6.9.8 Use of Polyacrylamide in Agricultural Soils

Farmers are increasingly using PAM in irrigated soils such as in western and northwestern USA (Fig. 6.10). The use of PAM amendments reduces soil erosion by about 1×10^6 Mg annually in these regions (Sirjacobs et al., 2000). The PAM use doubled between 1995 and 2005 in irrigated fields ($>200,000$ ha) for reducing furrow and sprinkler irrigation-induced soil erosion (Sojka, 2006). PAM can mitigate the erosion rates by as much as 95% and increase the infiltration rates by 15 and 50% in furrow-irrigated croplands, and application rates as low as 10 ppm (2 kg ha^{-1}) of

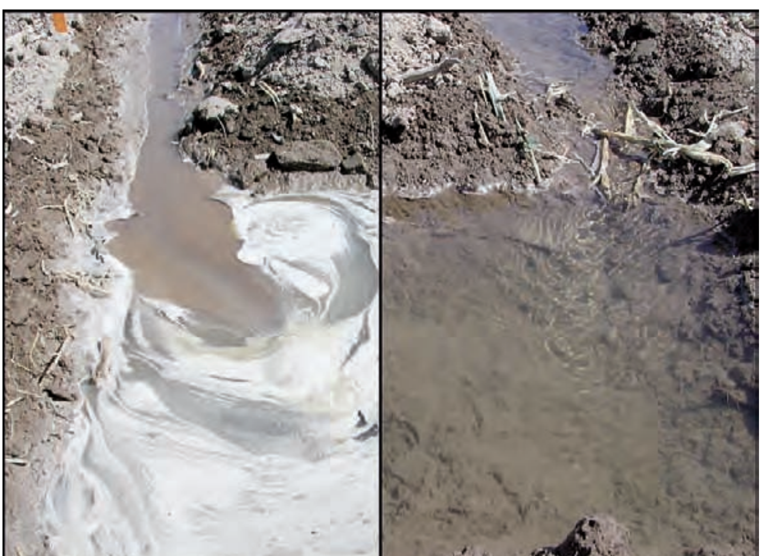

Fig. 6.10 Use of PAM in irrigation water reduces runoff sediment and soil erosion (Courtesy of the USDA-ARS, Northwest Irrigation and Soils Research Laboratory, Kimberly, ID)

PAM in irrigation water can provide sufficient erosion control (Sojka, 2006). PAM additions reduce soil erosion more than water runoff because PAM molecules are particularly effective in reducing soil detachment.

Optimum rate of PAM applications depend on site-specific characteristics. For example, application of 2–4 kg ha^{-1} of PAM can reduce soil erosion by 70–90% in some soils but only by 20% or less in others (Bjorneberg et al., 2000). On steep terrains and heavily irrigated soils, PAM application at rates of 20 kg ha^{-1} or even higher may be needed to effectively reduce soil erosion. A threshold level of application must be established for each soil. Too low or too high applications may not impact water infiltration rate and soil erosion control. Undissolved gels as a result of excessive PAM application reduce its effectiveness.

6.9.9 Use of Polyacrylamide in Non-Agricultural Soils

Apart from agricultural soils, PAM is also used to control soil erosion from urban areas, road cuts, landfills, and mined soils. Soil erosion from these disturbed sites can be as high as 160 Mg ha^{-1} yr^{-1} (Daniel et al., 1979). Intense rain storms between disturbance and vegetation cover establishment cause excessive erosion of soil from disturbed sites. Downstream water bodies (e.g. streams, lakes) adjacent to construction sites are often turbid due to heavy sediment input. Mulching, geotextile fabric covers, and dams are often used as temporary measures to control erosion from disturbed sites.

Use of PAM can be a short-term alternative to traditional erosion control practices. Unlike establishing a vegetation cover, PAM provides an immediate surface protection following disturbance when the soil is most vulnerable to erosion. Spraying PAM can even promote seed emergence and rapid plant establishment (Flanagan et al., 2002).

Use of PAM is often combined with that of gypsum to increase its performance. Rates between 20 and 80 kg ha^{-1} of PAM combined with 5 or 10 Mg ha^{-1} of gypsum can reduce erosion and water runoff by more than 50% in construction sites with steep slopes ($>$10%) (Flanagan et al., 2002). Use of PAM can reduce costs of traditional erosion control practices (e.g., mulch) in disturbed sites by more than 10 times. Polyacrylamide technology is a potential companion to other soil management practices for the rehabilitation and reclamation of degraded soils.

6.9.10 Cost-effectiveness of PAM

The low cost of PAM is becoming attractive to most landowners and farmers. The cost estimate for 1 kg of granular PAM is about $12 (Sojka and Lentz, 1997). The recommended rate of PAM per hectare for effective soil erosion reduction ranges between 4 and 20 kg depending on soil characteristics and severity of erosion. Thus, the cost of PAM technology for controlling soil erosion can be much lower compared to that of construction of difficult mechanical structures (e.g., sediment retention basins). Even use of mulch is about 12 times more expensive than that of dry PAM per hectare. Total annual cost for treating severely eroded soils with PAM may not exceed $160 per ha (Peterson et al., 2002). The need of repeated PAM applications for continuous soil erosion control particularly during peak rainy seasons may increase the total cost of PAM. The use of PAM alone or preferably in combination with other conservation practices can be a cost-effective approach to protect recently plowed or disturbed sites in sloping environments prior to vegetation establishment. The total cost of PAM use can be recovered by gains in soil and water conservation and crop yield improvements.

Summary

There are a number of biological and agronomic management practices to control water runoff and soil erosion including no-till, reduced tillage, crop rotations, cover crops, residue and canopy cover management, vegetative filter strips, riparian buffers, agroforestry, and synthetic conditioners. Canopy cover and surface residues are important determinants that influence soil erosion by intercepting raindrops and stabilizing soil surface. Soil erosion decreases exponentially with increase in canopy cover. Soil amendments such as animal manures, crop residues, and green manures are biological practices which reduce soil erosion and improve soil physical, chemical, and biological properties. There are numerous organic, natural, and synthetic amendments, each with specific attributes.

Cover crops, crop residues, and manure increase soil organic matter, increase water infiltration, and reduce runoff and erosion. Removal of residues for biofuel production can deteriorate soil properties, reduce soil organic matter concentration, alter water, air, and heat fluxes, reduce grain and biomass yields, accelerate soil erosion, disrupt nutrient cycling, and increase risks of non-point source pollution. Threshold levels of residue removal must be determined for each soil type and ecoregion prior to planning for large scale harvesting of crop residues. The amount of residue that can be removed as biofuel feedstocks varies among soil types and management systems. Bioenergy plantations are a viable alternative to removing crop residues from agricultural soils. Warm season grasses (e.g., switchgrass, miscanthus) and short rotation woody perennials (e.g., willow, poplar) can be grown on marginal soils to reduce the competition for land with food crops.

Soil conditioners such as PAMs with high molecular weights are also important to stabilizing soil and reducing soil erosion particularly in irrigated ecosystems. More than 400,000 ha of irrigated soils in the USA are treated with PAMs. Area of soils treated with soluble-PAM is the largest in Idaho. Polyacrylamides stabilize soil aggregates, improve soil surface roughness, increase the cohesiveness of soil particles, decrease aggregate slaking and detachment, reduce surface sealing and crusting, and flocculate suspended soil particles. PAM is a cost-effective practice to most landowners and farmers.

Study Questions

1. Does crop residue removal increase or decrease net greenhouse gas (CO_2, CH_4, and N_2O) emissions from no-till systems.?
2. What is the impact of: (1) leaving crop residues on the soil surface, and (2) plowing under the residues on soil physical, chemical, and biological properties and crop yields.
3. Describe the line-transect method for determining the percentage of residue cover on a given soil.?
4. What are the possible reasons for the more rapid impact of removing crop residues on silt loam soils compared to that on clayey soils.?
5. Describe the mechanisms responsible for the reduction of soil erosion by adding animal manure to the soil surface?
6. What are the main factors that improve performance of polymers for controlling soil erosion.?
7. Assume that PAM is to be sprayed on 1.5 ha of disturbed field in the form of a solution with concentration of 100 mg L^{-1}. What is the amount of water needed and the depth of water applied if the recommended rate of granular PAM for the entire field is 20 kg ha^{-1}?
8. What would be the longevity of PAM applied in Prob. 7.?
9. How do you determine the molecular weight and charge density of polymers.?
10. In what soils is the PAM most effective in controlling soil erosion.?

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Chapter 7
Cropping Systems

A cropping system refers to the type and sequence of crops grown and practices used for growing them. It encompasses all cropping sequences practiced over space and time based on the available technologies of crop production (Table 7.1). Cropping systems have been traditionally structured to maximize crop yields. Now, there is a strong need to design cropping systems which take into consideration the emerging social, economical, and ecological or environmental concerns. Conserving soil and water and maintaining long-term soil productivity depend largely on the management of cropping systems, which influence the magnitude of soil erosion and soil organic matter dynamics. While highly degraded lands may require the land conversion to non-agricultural systems (e.g., forest, perennial grass) for their restoration, prudently chosen and properly managed cropping systems can maintain or even improve soil productivity and restore moderately degraded lands by improving soil resilience. Crop diversification is an important option in sustainable agricultural systems (Table 7.1).

Management of cropping systems implies management of tillage, crop residue, nutrients, pests, and practices for soil conservation (Table 7.1). For example, excessive use of chemicals (e.g., fertilizers) for growing crops, particularly in developed

Table 7.1 Components of cropping systems

Tillage system and residue management	Cropping systems	Nutrient and water management	Erosion control practices
• No-till • Chisel tillage • Mulch tillage • Strip tillage • Residue removal • Residue burning • Partial residue removal • Quality of residues	• Fallows systems • Monoculture • Strip cropping • Multiple cropping • Contour strip cropping • Crop rotations • Cover crops • Mixed and relay cropping • Organic farming	• Precision farming • Use of amendments (e.g., manure, compost) • Enhancement of biological N fixation (BNF) • Irrigation/drainage practices • Water harvesting	• Conservation buffers • Windbreaks and buffer strips • Terraces and engineering devices • Sedimentation basins

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countries, has raised concerns over increasing risks of non-point source pollution. Discriminate use of inorganic fertilizers and other agrichemicals through precision farming and choice of appropriate cropping systems are useful strategies to minimize environmental pollution. Adopting organic farming, proper residue management, and complex crop rotations are examples of viable alternative cropping and management systems to conventional practices. The best combination of cropping practices for soil conservation must be determined for each soil and ecosystem. While there is a continued pressure for producing more food especially in developing countries of sub-Saharan Africa and South Asia, negative impacts of some cropping systems (e.g., monocropping) on quality of soil and water resources have raised some concerns. Cropping systems that are socially acceptable, economically profitable, and ecologically and environmentally compatible, and politically permissible must be designed for each ecosystem. The goal of a cropping system must be to conserve soil and water and sustain crop production.

7.1 Fallow Systems

Fallow systems consist of leaving a cropland either uncropped, weed-free or with volunteer vegetation for at least one growing season in order to control weeds, accumulate and store water, regenerate available plant nutrients, and restore soil productivity (SSSA, 2006). Systems based on plowed fallow are highly susceptible to wind and water erosion especially in the absence of volunteer or seeded vegetation. Bare fallow lands are either plowed or treated with chemicals to keep the land free of weeds and pests. These cultural operations, however, exert adverse impacts on soil quality. First, intensive plowing degrades soil structure, accelerates organic matter decomposition, reduces water infiltration, and increases soil erosion hazard. Second, pesticide use increases concerns about water pollution. Soils under continuous cultivated fallow systems have lower soil organic matter content and saturated hydraulic conductivity and higher runoff rates than those under no-till (Blanco-Canqui et al., 2004). Reduction in saturated hydraulic conductivity can increase runoff rates in fallow lands. Crop rotations that include long-term fallowing without vegetation cover reduce aggregate stability and nutrient concentration as compared to those that encompass a vegetation cover (e.g., forage legumes) during the fallow periods (Blair et al., 2006). Growing grass and legumes in place of bare fallow rotations is useful to providing permanent vegetative cover to soil and improve soil biological activity and nutrient cycling.

7.2 Summer Fallows

Summer fallow, without growing a cover crop, is a common fallow practice to store and conserve part of rainwater particularly in dry regions, in which evapotranspiration exceeds precipitation. Dryland farmers, such as those in western U.S. or Great

Plains, often rely on summer fallow to build soil water for winter wheat. Summer fallowing reduces water loss from plant transpiration, and water stored is used by the succeeding crop. Although the practice of summer fallowing has decreased in recent years, there are still about 20 Mha of summer fallow land in the USA mostly in the Great Plains and 6 Mha in Canada (Campbell et al., 2005). Regional and local climate (e.g., temperature, wind velocity, plant transpiration rate) and soil (e.g., texture, drainage, soil slope) conditions determine the length and frequency of summer fallowing. Fallow systems with rough soil surface and favorable soil structure are appropriate to absorb and retain water and reduce runoff. Because lands under summer fallow remain bare, proper management is crucial to reduce losses by excessive runoff and erosion. Plowing a fallow land is necessary to kill weeds and create rough surface for water storage, but its frequency and intensity must be minimized to reduce risks of soil erosion (Peterson and Westfall, 2004).

One of the conservation practices that has potential to replace summer fallowing is no-till farming, which not only conserves soil water but also increases organic matter pools as compared to fallowing. It maintains abundant crop residues on the soil surface, reduces soil evaporation, and increases soil water content in the root zone. Conversion of plow tillage to no-till reduces the need of summer fallowing and increases cropping intensity. It is economically profitable because it allows the production of more crops on the same piece of land and decreases use of C-based input. Intensification for cropping systems with the introduction of no-till and reduced tillage in wheat-summer fallow systems has improved precipitation capture and water storage and reduced soil degradation as compared to plowed summer fallows.

Higher return of crop residues in no-till soils also increases macroaggregation and total soil porosity. Increase in soil pore space captures more rainwater while increase in soil organic matter improves the soil's capacity to retain water. In a semiarid region of Spain, use of no-till in cereal-fallow rotations with 17–18 mo of fallow period proved to be the best strategy to protect the soil against erosion (Lopez et al., 2005). In some soils, yields from intensively managed no-till crops may be lower than those from systems with summer fallows. Yields from summer crops replacing fallows may, however, offset the differences. No-till crops leaving large amounts of residues are viable alternatives to fallow systems.

7.3 Monoculture

Monoculture refers to a cropping system in which the same crop is grown in the same field on a continuous basis. It is the single most common cropping system throughout the world principally in large-scale or industrialized farming. Monocropping makes planting and harvesting easy, but it makes the soil susceptible to erosion hazard, weed invasion, and pest and disease infestation (Table 7.2). It requires a periodic application of synthetic chemicals to supply nutrients and combat diseases with the attendant negative impacts on water quality. Monocropping with intensive tillage that leaves soil bare following harvest exacerbates soil erosion and eliminates crop and biological diversity.

Table 7.2 Implications of monocropping when managed under conventional tillage

Disadvantages	Advantages
• Eliminates crop diversity	• Allows specialization in a specific crop
• Degrades soil structure	• Favors large-scale farm/modern operations
• Reduces biological diversity	• Generates large volume of specific farm products and often produces higher profits
• Increases use of inorganic fertilizers and pesticides	• Reduces the cost of farm equipment
• Decreases crop yields	• Makes seed preparation, planting, harvesting relatively simple
• Increases soil's susceptibility to erosion, weed invasion, and pest incidence	• Reduces cultural operations
• Decreases soil resilience	• Narrows harvesting times
• Decreases wildlife habitat	• Increases profit due to economy of scale

High demands for specific products have spurred large-scale monocropping. The resultant lack of crop diversification reduces soil biological diversity, wildlife habitat, and soil resilience. The number of main crops in the world has been reduced to <12 and only four crops (rice, wheat, corn, and potato) predominate (Esquinas-Alcázar, 2005). Presently, monocrops occupy more marginal and degraded lands in the world resulting from both the degradation of prime agricultural land and expansion of monocropping. Studies in Ghana have reported that maize yields and nutrient accumulation were larger in maize/cowpea rotation than maize monocropping (Horst and Hardter, 1994).

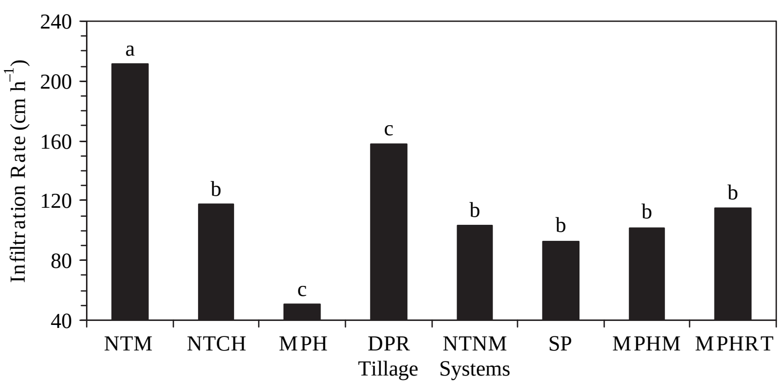

Fig. 7.1 Impacts of corn monocropping on water infiltration rates under different tillage systems (After Lal, 1997). NTM = No-till + mulch; NTCH = NT + chiseling; MPH = Moldboard plow + harrowing; DPR = Disc plow + rotovation; NTNM = No-till + no mulch; SP = Summer plowing; MPHM = Moldboard plow + harrowing + residue mulch; MPHRT = Moldboard plow + harrowing + ridge till. Bars followed by the same lowercase letter are not significantly different ($P < 0.05$)

In western Nigeria, 8-yr monocropping of corn reduced crop yields and deteriorated soil physical properties, and the negative impacts of monocropping were more severe under plow till than under no-till farming. The water infiltration under no-till management tends to be higher than under plowed soils (Fig. 7.1). The magnitude of adverse impacts of monocropping on soil function depends on soil, tillage system, and climate. No-till monocropping is more sustainable than monocropping under plow tillage. On a Rayne silt loam in Ohio, 42-yr no-till continuous corn with manure maintained or even improved soil physical properties, and sustained crop production compared to the adjacent moldboard plowed continuous corn without manure (Blanco-Canqui et al., 2005).

7.4 Crop Rotations

Crop rotations are systems in which different crops are grown sequentially on the same field in alternate seasons or years. Switching crops in a recurring fashion under a planned sequence contrasts with continuous monoculture. Planting three or more different crops before returning to the original crop constitutes long-term rotations. The larger the number of crops involved in a rotation, the greater the benefits to soil productivity and plant diversity. Crop rotation is one of the simplest and the most desirable strategies of soil and water conservation. There are three main types of rotations based on the duration (Karlen et al., 1994):

1. ***Monoculture***. It is confined to a single crop with no diversity.
2. ***Short rotation***. It is basically a 2-yr rotation (e.g., corn-soybean).
3. ***Extended rotation***. It refers to >2-yr rotations (e.g., corn-oat-wheat-clover-timothy).

Based on the crop and plant species used, crop rotations are classified as:

1. ***Annual***. It refers mostly to monoculture (e.g., corn).
2. ***Annual-perennial***. It includes rotations with row crops and perennials (e.g., corn-alfalfa)
3. ***Diverse***. It includes more than three crops (e.g., corn-oats-wheat-hay).

Rotating different crops is an ecologically viable alternative to monocropping and is relevant to addressing agricultural and environmental concerns. Long rotations are preferred over monocropping and short-rotations. Economic pressures have led to monocropping or short rotations such as is the case in the U.S. Corn Belt region where rotations are commonly confined to corn-corn or corn-soybean. About 80% of corn and soybean in this region is either under monocropping or in rotation with these two crops (Allmaras et al., 1998). Monocropping with corn occupies <25% of cropped land in midwestern U.S. states.

Extended crop rotations are useful practices to conserve soil and water and sustain agricultural production (Fig. 7.2). Short-rotations, depending on the crops, may not be any better than monocropping for conserving soil and water. Indeed, soil

Fig. 7.2 Corn-alfalfa rotation to conserve soil and improve soil fertility in central Ohio (Photo by H. Blanco)

erosion rates from intense short-rotations with corn and soybean can be equal to or surpass those from monocrops (Van Doren et al., 1984). Small scale farmers have traditionally practiced diversified cropping systems. Before 1940's, use of extended crop rotations was high. Agricultural mechanization, large-scale farming, availability of heavy farm equipment, intensive use of fertilizers and pesticides, and high economic returns have all favored short rotations and monocropping.

Benefits of crop rotations can not be, however, overemphasized because they:

1. reduce soil erosion,
2. improve soil properties,
3. increase organic matter content,
4. improve soil fertility,
5. increase crop yields,
6. reduce build-up of pests,
7. increase net profits,
8. improve wildlife habitat,
9. reduce use of chemicals, and
10. reduce water pollution.

7.4.1 Soil Erosion

Rotations which include high above- and below-ground biomass producing forages and crops reduce soil erosion hazard. Growing cereals and legumes alternatively with row crops provides a dense and permanent vegetative cover that stabilizes the soil underneath, reducing soil erosion hazard. In regions with high potential of water and wind erosion, short rotations with row crops are not sufficient to reduce soil erosion to minimum levels. Alternatively, rotating row crops over longer time intervals (>2 yr) with legumes and perennial grass for hay and pasture is an effective soil conservation practice. Compared to continuous row crops, rotations with hay or pasture systems reduce soil erosion by 80–90%, while short rotations reduce it by <30% compared to monocropped systems (Jankauskas et al. 2004). Incorporation of wheat into rotations with corn or soybean reduces soil erosion more than corn-soybean rotations alone (Karlen et al., 1994). Rotations with diverse forage and grain crops in association with other soil conservation practices must be established in soils where erosion risks are severe. In Indonesia, long-term cropping of cassava produced higher soil erosion and lower economic returns than rotations of cassava-corn-soybean-cowpea (Iijima et al., 2004).

The long-term (>100 yr) crop rotation experiments in Sanborn Field, one of the oldest agricultural research fields in the world established in 1888 at the University of Missouri, Columbia, illustrates the distinct benefits of crop rotation management to reducing soil erosion risks. In this centennial field, a 6-yr crop rotation with corn-oat-wheat- clover-timothy reduced topsoil loss and maintained the profile soil textural characteristics as compared to continuous monocropping with corn after 100 yr (Gantzer et al., 1991). The topsoil thickness in continuous corn was only about 60% of that in rotation plots due to water, wind, and tillage erosion. Continuous timothy reduced topsoil losses to negligible levels even when compared with other rotations, which suggest that continuous grass cover is one of the best management options to control soil erosion. The lower soil erosion hazard with crop rotations is due to improved soil stability against slaking and detachment, which are critical processes of soil erosion. At Sanborn Field, the percentage of water-stable aggregates under continuous corn was about 70% of that in corn-wheat-red clover rotations, while soil splash under continuous corn was twice as much as that under rotations, portraying the high susceptibility of monocropped soils to erosion (Rachman et al., 2003).

7.4.2 Soil Physical Properties

Rotating with crops characterized by high above- and below-ground biomass production plant species reduces soil bulk density, increases aggregation, improves soil macroporosity, and stabilizes soil. Improvements in soil structural stability occur when rotations are used in combination with no-till farming as a result of positive interactions between crop diversity and absence of soil disturbance. Plant available water content is higher in no-till rotation systems compared to conventionally managed monocropped systems. Crop rotations that include deep-rooted legumes also increase water movement in the soil profile. Across a wide range of soils with differing texture and drainage conditions in Minnesota, saturated hydraulic conductivity under diverse crop rotations including corn-soybean-alfalfa-small grain was higher than that under 2-yr corn-soybean rotations (Oquist et al., 2006). The degree of improvements in soil properties caused by rotations depends on the amount of residue left after harvest and the root biomass. Crops such as corn leave more residue than soybean and, thus, protect the soil against from erosive energy of raindrops and crusting. Crop rotations reduce bulk density and increase aggregate stability in contrast to monocrops (Karlen et al., 2006).

7.4.3 Nutrient Cycling and Input

Crops vary in their ability to absorb, maintain, and supply nutrients. While row crops (e.g., corn) extract and reduce most of the essential nutrients in the soil, combination of corn with legumes (e.g., soybean, alfalfa) reduces N losses. Rotations with legumes have the ability to enhance microbial activity, fix atmospheric N, and supply non-synthetic N to succeeding crops. Crop rotations also reduce the loss of nutrients by reducing soil erosion. In essence, long rotations improve nutrient cycling and storage by: (1) supplying nutrients, (2) reducing nutrient loss in runoff, and (3) improving soil biological activity. These beneficial effects of legumes persist for two or three yr following legume cultivation. Using sod- and bunch-grass in rotations is a strategy to increase soil organic matter content because of high above- and below-ground biomass input. The abundant biomass and deep growth pattern of grass roots absorb nutrients from deeper soil, promote microbial processes, and increase nutrient cycling. Crop rotations that leave abundant residues on the soil surface after harvest are particularly important to recycle and build organic matter and nutrients in the reserves.

7.4.4 Pesticide Use

Infestation by insects, nematodes, diseases, and weeds is specific to a crop. Thus, rotating crops interrupts and eliminates the pest cycles and reduces the use of pesticides. The reduction in use of pesticides and synthetic fertilizers with crop rotations results in less non-point source pollution. The effectiveness of crop rotations for controlling pests depends on the nature and specificity of pests. Rotations are effective measures whenever the pests are: (1) specific to a crop and field, (2) not widely spread across crops, and (3) do not increase under the absence of host crops. Insects such as corn rootworm, wheat stem sawfly, wheat stem maggot, Hessian fly in wheat, alfalfa weevil, sweetclover weevil, and sugar beet maggot, and root aphid are effectively controlled by switching host crops (Bauder, 1999). Corn-soybean rotations in the U.S. Corn Belt region have been a good deterrent against corn rootworms because the eggs laid under corn typically hatch during the next spring when the land is under soybean. Pesticides applied to control corn rootworm in USA represent about 20% of total pesticide applications (Pikul et al., 2005). Extended rotations are more effective at reducing corn rootworm attacks than short rotations. For example, 2-yr corn-soybean rotations may not be sufficient to break the insect life cycles.

7.4.5 Crop Yields

One of the immediate and direct benefits of crop rotation is the increase in crop yields (Bauder, 1999). For example, corn grown after alfalfa or soybean often produces higher yields than continuous corn systems. Pikul et al. (2005) reported that corn grain yield was 6.1 Mg ha^{-1} under corn-soybean rotations, 7.3 Mg ha^{-1} under corn-soybean-wheat/alfalfa-alfalfa rotations, and only 3.83 Mg ha^{-1} under continuous corn in systems without N fertilization. Differences in corn yield among the three cropping systems were not, however, significant when high rates of N fertilizer were applied. While monocropping tends to maximize crop yields through the application of fertilizers and pesticides, the practice of rotations with legumes reduces the use of N fertilizers. Crop rotations reduce production costs and increase net profits by increasing crop yields and by reducing inputs (e.g., fertilizers, pesticides). Rotating crops every year also adds diversity to the system and flexibility against price fluctuations. Crop rotations adopted in conjunction with no-till agriculture save energy by elimination of tillage. Economic benefits are often more in longer than in shorter rotations ($>$3 yr).

7.4.6 Selection of Crops for Rotations

The selection of crops for a rotation sequence varies with local and regional characteristics. It depends on the soil type, soil fertility, soil slope, economic and market goals, presence of pests, and livestock type. In the midwestern U.S., 2-yr corn and soybean rotation has become a popular practice since 1950’s. This relatively new rotation structure has somewhat replaced more diverse rotations which included oats, wheat, and alfalfa. Decrease in livestock has reduced demands for oats and alfalfa, and similarities in farm equipment, cultural operations, growth requirements, labor costs, economic profits, marketing options, and numerous food and industrial uses of corn and soybean have triggered the expansion of this rotation (Karlen et al., 2006). Implications of corn-soybean rotations on soil and water quality, agricultural sustainability, crop diversity, and environmental quality are, however, questionable. Conventionally tilled large-scale corn-soybean rotations degrade soil structural properties. As an alternative, rotations including more than two crops are proposed to improve diversity of food products, enhance biological activity, and build resistance against pest incidence.

Crop rotations that include alfalfa, clover, or perennial grasses are recommended to improve soil structure, macroporosity, reduce soil compaction, and increase soil organic matter content. Growing perennial crops in rotation with row crops eliminates tillage and reduces wheel traffic. Deep-rooted ($>$1 m) legumes or grass species loosen relatively compact or impermeable soil horizons, ameliorate plowpan formation, improve soil porosity, promote infiltration rate, and reduce runoff and soil erosion. Proliferation of roots and reduced soil disturbance under perennial crops promotes soil aggregate stability and strength. Inclusion of perennials in traditional crop rotations improves soil fertility over rotations with summer annuals only.

In the highlands of Ethiopia, combined management of intercropping wheat with clover and rotation with oat-vetch-chickpea significantly has been used as a successful alternative for producing high quality fodder (Tedla et al., 1999). Monocropping of cereals generally produces lower grain yields than legume-cereal rotation. In Lituana, replacing potatoe-barley-rye-clover-timothy rotations with perennial grass species including red fescue, white clover, Kentucky bluegrass, and birdsfoot trefoil in fields with $>10\%$ slope gradient reduced soil losses from 14.5 to $0\,m^3\,ha^{-1}$ (Jankauskas et al., 2004). In essence, multi-species legume and grass species must be incorporated in row crop systems to rejuvenate soil and reduce its erodibility because row crop rotations are not sufficient to reduce soil erosion to tolerable levels in highly erodible soils. Indeed, crop rotations perform poorly in saline, sandy, and highly erodible soils unless used in conjunction with other conservation measures.

7.5 Cover Crops

Benefits of cover crops are discussed in more detail in Chapter 6. Cover crops are an integral component of cropping systems to conserve soil and water. They protect soil against erosion, improve soil structure, and enhance soil fertility. Cover crops with legumes and mixture of plants enhance performance of crop rotations. In the U.S., common winter cover crops used in rotation cycles include rye, clover, and vetch (Lal, 2003). Crop rotations and cover crops are effective conservation practices. Both are grown to benefit the soil and optimize crop yields in a way that is best suited to a specific land. A well-structured system with cover crops and rotations restores soil productivity. Legume cover crops enhance biological nitrogen fixation and biomass input. When used synergistically, crop rotations in conjunction with cover crops reduce incidence of insects and weeds and diseases, improve soil productivity, and accentuate sustainability and profitability.

7.6 Cropping Intensity

Cropping intensity is the ratio of total cropped or harvested land over total cultivated or arable land over a specific period of time.

$$\text{Cropping Intensity} = \frac{\text{Total cropped land}}{\text{Total cultivated land}} = \frac{\text{Number of crops}}{\text{Unit of land}} \tag{7.1}$$

Cropping intensity refers to the number of crops grown on the same piece of land in a specific time period (e.g., 2 yr). Cropping systems that favor intensive cropping produce more biomass and provide higher plant diversity resulting in better soil condition for crop production than less intense systems. Reducing fallow (e.g., summer fallows) frequencies and planting multiple crops in rotation are examples of intensive cropping. Continuous tillage, extended fallow periods, and reductions in cropping intensity and diversity lead to soil degradation.

7.7 Row Crops

Row crops refer to crops grown in parallel rows (Fig. 7.3). These crops are usually profitable, representing a significant portion of world agriculture. Corn, wheat, rice, soybean, cotton, peanuts, sorghum, sugarcane, sugar beets, and sunflowers are examples of row crops. Soil erosion is a major concern in intensive row cropping systems under plow tillage system of seedbed preparation. The unprotected wide space between rows exacerbates risks of rill and gully erosion. Corn and soybean are usually planted in rows spaced 0.76–1 m apart although row spacing of <0.75 is recommended, and can be as narrow as 0.36 m. Reducing space between rows has important implications to soil and water conservation and crop yields. Crops grown in narrow spaced rows provide better protection against raindrop impacts by forming a close canopy. Higher canopy cover or closed canopy cover in narrowly spaced row crops as compared to wide rows reduces evaporation and decreases soil's susceptibility to erosion. The closed canopy cover rapidly shades the soil surface, reduces soil temperature and weed proliferation although vehicular traffic can be difficult. Mechanical operations for plowing between rows and herbicide application require relatively wide row spacing. In terms of crop yields, effects of row spacing are often inconsistent. Narrow row spacing (<0.75 m) may increase crop yields than wide spacing in some soils while have no effect in others (Lambert and Lowenberg-DeBoer, 2003).

Incidence of weeds and insects is affected by row spacing. Reducing corn row spacing can increase attacks, for example, of corn rootworm larvae on root growth (Nowatzki et al., 2002). Economic risks and equipment costs (e.g., equipment consolidation) for changing row spacing must be assessed against soil and water conservation benefits. Before 1930s, row spacing was determined by animal-drawn

Fig. 7.3 Row crops involving onion (*left*) and corn (*right*) with little or no residue cover (Photo by H. Blanco). The bare interrows with wide spacing can develop rills

equipment and was often preset at about 1.1 m. The advent of tractors in the 1950s has made possible reducing row spacing from 1.1 to 0.75 m while increasing crop yields by about 10%. Further reductions in row spacing down to 0.36 m have not always increased crop yields, depending on crop varieties, cropping system, and site-specific conditions. Reducing row spacing can increase costs of production by modifying combine heads, tractors (e.g., tires, rims), and planters. Application of herbicides and fertilizers also increases in narrow rows as chemicals are applied on the basis of amount per row width. From the soil and water conservation perspective, narrow rows are preferred because of the protective effect of increase in the canopy cover.

7.8 Multiple Cropping

Multiple cropping is a system where a single crop species is grown more than once or different crops are simultaneously planted on the same field during the same season a year. It is a popular practice among small farmers in developing regions (e.g., Africa) because it allows an integration of food crops, farm animals, conservation grass buffers, and trees into the same piece of land. Planting several crops extends the harvest season either with earlier or later ripening crops while providing greater vegetative surface cover and diverse crop produce over a long period of time. Under appropriate climatic (e.g., water supply) and soil conditions, multiple cropping is a source of year-round supply of grains, fruits, and vegetables. The advantage of multiple cropping is that it comprises all the interactive variables and factors of different plants and the environment. The number, selection, and combination of crops (e.g., corn, soybean, vegetables) depend on local soil, climate, and ecosystem conditions.

Multiple cropping is advantageous because it:

- allows the production of diverse food crops,
- offers better soil erosion control by continuous growing of crops with variable biomass production and rooting systems,
- reduces risk of total loss of crops from adverse climate conditions (e.g., drought resistant) or diseases,
- provides diversified farm products from a small piece of land, reducing production costs.
- improves soil fertility and reduces soil erodibility by planting grass, grain crops, and legumes,
- reduces disease pressure and use of synthetic fertilizers, herbicides, and pesticides by dense planting and intensive management, and
- allows planting crops in different seasons, spreading the harvest and supply of produce.

In a few cases, multiple cropping may exacerbate pest invasion and survival because pests can move from one crop to another. Land fractionation in small plots may not accommodate mechanized farming and row crop planting with large farm equipment. Overall, multiple cropping is a more intensive management and more profitable farming system than single or one crop per year. Double cropping, intercropping, and relay cropping are among the most common multiple cropping systems.

7.9 Double Cropping

Double cropping consists of planting crops following harvest of the first on the same land during the same year. This practice thus consecutively produces two crops on the same land in one year. Harvesting wheat in early summer and planting corn or soybeans on the same land to be harvested in fall is a common example of double cropping in temperate regions. The three to four months of growing season remaining after wheat harvest leaves sufficient time for growing either corn or soybean as a second crop. Double cropping is suited to regions with long growing seasons. Depending on the ecosystem, double cropping increases profits by harvesting twice the same or different crops. The possible reduction in high yield by late planting of the second crop may be offset by the yield of the first crop or viceversa. In Missouri, wheat-amaranth, canola-amaranth, wheat-sunflower and wheat-soybean systems are the commonly used double cropping systems with highest net returns (Pullins et al., 1997). Double cropping with canola is often less profitable than with wheat, and sunflower planted after either canola or winter wheat was a viable alternative to soybean.

No-till management is compatible with double cropping as long as full season crop residue does not interfere with planting. Dense and abundant crop residues are important to reducing erosion and evaporation, but the thick mulch may make no-till planting in double cropping systems difficult. Nonetheless, double cropping is advantageous because it provides a protective vegetative cover all year long while improving farm income and breaking up pest cycles. Producing two crops in a single crop year is suited for both grain and forage production if managed properly. Summer annual grasses and perennial forage legumes can fallow winter wheat and used for livestock. Planting annual small grain or ryegrass following corn harvesting for silage in late summer or early fall soybean harvests is also an option.

7.10 Relay Cropping

Relay cropping consists of interseeding the second crop into the first crop before harvesting. It allows the production of a second crop during the same year. The same crop or different crops can be planted in relay cropping, which provides a continuous supply of food. In temperate regions, the second crop often follows winter wheat. Relay cropping is appropriate if: (1) there is sufficient time for the production of a second crop before the first frost, and (2) there is adequate soil water supply to sustain a second crop. Water availability is the main determinant of relay cropping. In soils with limited water holding capacity, relay systems rely on irrigation or adequate rainfall although irrigation increases the production costs. Relay cropping is difficult in arid and semi-arid regions due to limited supply of water.

7.11 Intercropping

Intercropping is a multiple cropping system where two or more crops are grown simultaneously on the same field. The different crops can be planted in alternating rows or sections. Intercropping mixes different plant species with contrasting height, foliage, biomass, and other agronomic characteristics. It is a recommended system for soil and water conservation. Additional weeding, difficult harvesting, and decreased crop yields may be among possible shortcomings of some intercropping systems. Intercropping takes into account all beneficial interactions between and among crops while creating possible negative interactions caused by the neighborly effects. It minimizes pest problems and improves soil fertility. For example, plant species such as garlic and onion repel certain insects and protect adjacent vegetables (e.g., tomato, lettuce, carrot) from pest attacks provided that the competition for light and water is negligible. Intercropping with legumes or deep-rooted plant species absorbs nutrients from deeper soil horizons and reduces N deficiencies among neighboring and succeeding non-legume crops. Fruit trees can be important components of the mosaic of multiple cropping. Intercropping with trees (agroforestry) allows planting annual crops between rows of trees and has multiple benefits.

7.12 Contour Farming

Contour farming is the practice of tilling, planting, and performing all cultural operations following the contour lines of the field slope. This practice contrasts with up- and down-slope farming, which is the least desirable practice on highly erodible sloping lands. Furrows in an up- and down-slope direction become channels of concentrated runoff, forming rills or even gullies. Contour farming is being adopted in modern agriculture across the world for soil erosion control. Contouring creates furrows perpendicular to the predominant field slope. These furrows retard the runoff velocity, reduce the runoff transport capacity, enhance water infiltratibility, reduce sediment transport, and discharge excess runoff at non-eroding velocities. Furrows on the contour create irregular field surface which reduces runoff velocity. Deep and permeable soils respond better to contouring.

Contour farming effectively reduces rate of erosion in soils with slopes of up to 10% (Fig. 7.4). On steeper slopes, contour cropping can still be used to control erosion but must be accompanied by other conservation practices such as grass waterways to safely discharge runoff water from the contour rows. In sloping soils in

Fig. 7.4 Contour farming reduces erosion and improves soil productivity in sloping fields (Courtesy USDA-NRCS)

China, contour cultivation on terraces is a common practice to conserve soil and water (Fullen et al., 1999). In a clayey and sloping soil in the Philippines, contour cropping, strip cropping, and hedgerows were all effective at reducing soil erosion, but contour cropping was the best (Poudel, 1999). Annual soil erosion were measured at 65.3 Mg ha^{-1} for up-and-down tillage, 45.4 Mg ha^{-1} for contour hedgerows, 43.7 Mg ha^{-1} for strip cropping, and only 37.8 Mg ha^{-1} for contouring (Poudel, 1999). Contour cropping in combination with reduced tillage and residue return reduces runoff and soil erosion and increases crop yields as compared to up- and down-slope tillage. Contour cultivation is an ideal conservation practice but its use on steep slopes and rolling topography ($>$20%) may be limited by the instability of farm machinery, which can slip down the steep slopes especially when the soil is wet.

7.13 Strip Cropping

Strip cropping refers to the practice of growing crops in alternate strips of row crops or forage/grass. This cropping system is an effective practice to reducing soil erosion because it breaks sloping landscapes in wide segments with diverse vegetative cover which intercepts runoff and promotes water infiltration, thereby reducing runoff and soil erosion. Strip cropping is often integrated with rotations where strips are planted to different crops each year. Hay, pasture or legume forages are also commonly used in strips in rotation with row crop crops. The sod or perennial grass is particularly effective at slowing runoff and filtering out sediment. Strip cropping established perpendicular to the dominant slope reduces soil erosion as compared to bare soil or up-down slope cropping or tillage. Crop yields between strip cropping and monocultures may not significantly differ in most cases, but the greatest benefit with strip cropping is to soil erosion control.

The width of the strips depends on soil slope, erosion potential, crop type, and equipment size. Narrow strips reduce flow lengths more effectively than wide strips. The width of strips must match the equipment turn or width for cultivation. On gentle slopes of up to 5%, a strip width of about 30 m is recommended, while on steeper slopes the width must be less than 20 m (Bravo and Silenzi, 2002). Strip cropping may also be used in nearly flat terrains to reduce wind erosion. Risks of water and wind erosion increase with increase in strip width. Proper spacing of strips is important to effectively reduce soil erosion. Poorly designed strips may actually increase runoff and soil erosion if they concentrate runoff and have sparse and temporary vegetative cover.

7.14 Contour Strip Cropping

This cropping system involves planting row crops in strips on the contour of the field slope (Fig. 7.5). It provides added erosion control and plant and crop diversity because it combines contour- and strip-cropping. Strip-cropping on the contour is more effective than contouring alone for reducing soil erosion in fields with severe erosion hazard. Contour strip cropping systems can reduce soil erosion to $<40\%$ as compared to systems without these practices or with contouring alone (Francis

Fig. 7.5 Contour stripcropping protects the soil from erosion and improves land aesthetics (Courtesy USDA-NRCS)

et al., 1986). When combined with high rates of crop residue return, soil erosion from these systems can be as low as 5% of the maximum. The grass, legumes or small grains used in strips slow runoff and trap sediment leaving row crops. Permanent grass/legumes strips must be maintained between strips in soils with severe erosion. The strips can be used as traffic lanes for cultural operations. The mixture of grass and legumes provides hay and benefits to wildlife habitat and plant diversity. Permanent strips also provide nesting, food, shelter to small animals.

7.15 Land Equivalent Ratio

Land equivalent ratio (LER) is an index of combined yields of different intercrops with respect to the yield of sole culture of the same crops (Francis et al., 1986). Determining the LER for a specific cropping system consists of summing up the ratios of intercrops or strip crops to the yields of sole crops to evaluate the overall efficiency of intercropping or strip cropping. The LER is a measure of productivity of intercropped systems. It estimates whether a strip crop is equal or more profitable than monocropping with the same crop once crop yields and production costs are weighed in. It is also used to estimate the land area required to grow crops in strips compared to the amount of land required to grow monocrops of each crop. An LER >1 means that intercropping or stripcropping is better than monocropping whereas LER <1 means the opposite. For example, an LER of 1.20 signifies that an area planted to a monocrop require 20% more land to produce the same yield as the same area planted to an intercrop or strip crops. An LER of 0.90 indicates that total intercrop yield was only 90% of the yield of the sole crop. The advantage of LER is that it measures the positive and negative interactions of intercropping systems.

For example, consider that corn yields $5\,\mathrm{Mg\,ha^{-1}}$ when grown alone and 6.5 $\mathrm{Mg\,ha^{-1}}$ when intercropped, and soybean yields $2.5\,\mathrm{Mg\,ha^{-1}}$ when alone and 2.0 $\mathrm{Mg\,ha^{-1}}$ when intercropped. These figures mean that, under corn-soybean intercropping, half hectare of corn would yield $3.25\,\mathrm{Mg\,ha^{-1}}$ or 65% of the sole crop while the remaining half hectare of soybean would yield $1.0\,\mathrm{Mg\,ha^{-1}}$ or 40% of a sole crop. For this system, the LER would be:

$$\mathrm{LER} = \frac{\text{Strip Crop1}}{\text{Monocrop1}} + \frac{\text{Strip Crop2}}{\text{Monocrop2}} + \frac{\text{Strip Crop3}}{\text{Monocrop3}} + \ldots\ldots\ldots\ldots + \frac{\text{Strip Crop}_n}{\text{Monocrop}_\mathrm{n}} \quad (7.2)$$

$$\mathrm{LER} = \frac{3.25}{5} + \frac{1}{2.5} = 0.65 + 0.4 = 1.05$$

The LER shows that strip cropping is 5% more efficient than monocropping. The LER value of 1.05 also shows that 5% more land would be needed to obtain the same amount of yield in corn and soybean monocropping. Hauggaard-Nielsen et al. (2003) reported on a temperate sandy loam that pea-barley intercrop yielded $4.0\,\mathrm{Mg\,ha^{-1}}$, $0.5\,\mathrm{Mg}$ lower than the yields of monocropped pea, and about

1.5 Mg ha^{-1} higher than monocropped barley. The LER value also showed that intercropping used nutrients 17–31% more efficiently than monocropping and N fixation increased from 70 to 99% by intercropping. The LER for corn-soybean strip cropping for the midwest U.S. ranges between 0.95 and 1.15 (Francis et al., 1986). Differences in strip width, tillage management, soil texture, land slope, and number of crops explain the inconsistencies in LER values. Overall, strip cropping is preferred over monocropping because it reduces soil erosion, improves biological diversity, and rejuvenates soil fertility. Long-term economic gains and maintenance of soil productivity are sufficient reasons to adopt strip cropping systems.

7.16 Organic Farming

Obtaining high crop yields to meet the increasing demands for food and fiber has been equated with intensive tillage, accelerated mechanization, high chemical input, and use of genetically engineered crop varieties (e.g. hybrids) particularly in developed countries. The conventional way of improving soil fertility is through the addition of highly soluble inorganic fertilizers. Likewise, combating pests and diseases has heavily relied upon frequent and high input of commercial pesticides. The development of relatively inexpensive inorganic fertilizers and pesticides has contributed to the expansion of chemically-based agricultural production systems resulting in large increase in cultivated land area and crop yields. While conventional farming systems have revolutionized agriculture, these have also created major problems about non-point source pollution, decline in biodiversity, and increase in soil degradation. Thus, the challenge lies in developing an alternative system that reduces or eliminates input of chemicals while sustaining high crop yields. One of such potential alternatives is organic farming.

7.16.1 Definition

Organic farming is an agricultural system where no synthetic fertilizers or pesticides are used to produce food and fiber in contrast to chemically-based conventional farming systems. It is also called biological or biodynamical agriculture because it improves soil biology, enhances soil's natural fertility, and promotes plant biodiversity. It is a system that comprises a host of environmentally friendly agricultural practices to sustain crop production. Organic fertilization to add nutrients and mechanical and biological practices to control pests are two key exclusive components of organic farming (Reganold et al., 1987). Crop rotations, cover crops, manuring, residue mulch, and compost are among the alternative sources of nutrients used in organic farming. Organic farming encompasses all crops (e.g., grains, cotton, vegetables, flowers), and animal products (e.g., meat, dairy, eggs) and processed foods.

7.16.2 Background

Organic farming dates back to the origins of agriculture. Prior to the advent of modern agriculture, neither fertilizers nor pesticides nor herbicides were used to produce crops. Small-scale and traditional farmers relied solely on organic amendments such as animal manure to fertilize their fields. Weeds have been traditionally controlled by manual operations. Tractors were not yet available, and soil disturbance was minimum. Thus, most of the farming systems in the pre-modern era would have been regarded as organic farming combined with reduced tillage.

The boom of highly mechanized agriculture and fertilizer (e.g., N) industries following World War I (1930's) changed the paradigm of agriculture. It dramatically increased both chemical use and crop yields particularly in the developed world (Lotter, 2003). Nearly at the same time, concerns over excessive use of synthetic fertilizers resulted in the emergence of organic farming in Europe (Germany, England, and Switzerland). While some regard the present-day organic farming somewhat a resemblance of pre-modern agriculture, there are substantial differences in management. Current organic farming systems use intensive mechanized tillage to control weeds and require the certification detailing the cultural practices, commercialization of products, and establishment of conservation practices for each farm. Certified organic refers to products grown and processed based on strict compliance with standards of organic farming. The number of certified organic farms is rapidly increasing and is mostly (60%) used in vegetable production (Willer and Yussefi, 2004). The total cultivated land under organic farming increased linearly around the world between 2000 and 2006 (Fig. 7.6), and is projected to increase to 10% in the USA and 20–30% in Europe by 2010 (Lotter, 2003). In terms of percentage of total land used for cultivation of organic crops (e.g., vegetables), Europe is the first followed by Latin America (Fig. 7.7).

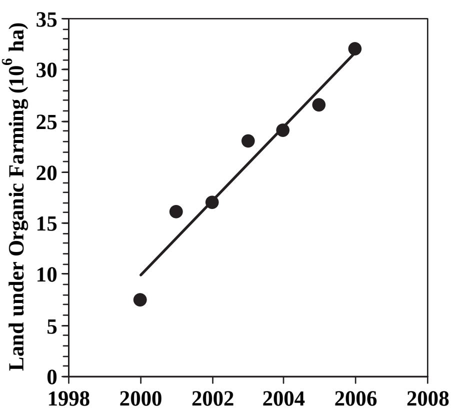

Fig. 7.6 Increase in organically farmed land in the world between 2000 and 2006 (After Willer and Yussefi, 2004)

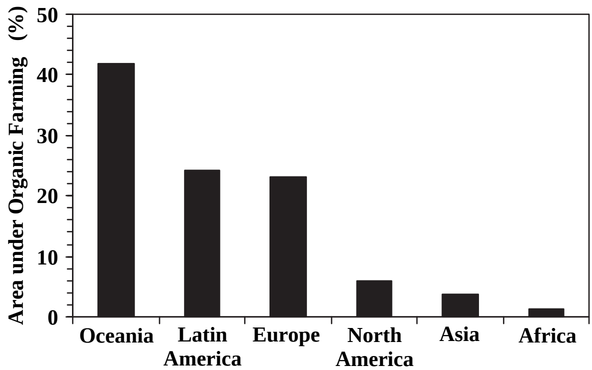

Fig. 7.7 Distribution of percentage area under organic farming of total land used in the world (After Willer and Yussefi, 2004)

7.16.3 Importance

Organic farming mimics the natural environment and builds soil organic matter content. The goal of organic farming is to maintain a diverse and active ecosystem of soil organisms for replenishing nutrients, improving/maintaining soil properties, and promoting biological diversity while ensuring a sustained crop production. Conventional farming produces abundant and low cost food at the expense of soil deterioration and environmental pollution, and its long-term production is thus questionable. In contrast, organic farming provides many benefits over conventional farming (Table 7.3). Increasing demands of organically grown produce makes organic farming an economically viable system. With the advent of transgenic crops and processed foods with chemical additives, there is an ever growing interest in organic farming. The increase in interest is driven by increasing; (1) demands for high quality food products, (2) concerns of environmental pollution, and (3) environmental regulations.

Marketing of organic foods is progressively expanding. In the USA, sales of organic food increased by about 20% between 2000 and 2007. In some European countries, financial subsides are provided by the government to promote and make organic farming more competitive (Siegrist et al., 1998). About 8% of the cultivated land area in Europe is under organic farming (Mader et al., 2002). Research on organic farming and marketing of products is rapidly advancing.

7.16.4 Water Quality

The greatest advantage in adopting organic farming is the improvement in water quality. Conventional farming systems, based on high input of chemicals, have caused pollution of streams, rivers, and lakes. Synthetic nutrients and pesticides are soluble and are rapidly transported in runoff and seepage to surface and ground waters. Elevated concentrations of agrichemicals in coastal waters (hypoxia) such as in the Gulf of Mexico question the long-term sustainability of conventional farming systems.

Table 7.3 Potential benefits of organic farming over conventional farming

Conventional farming	Organic farming
• Produces rapid and high volumes of food crops	• Produces often low but sustained food crops
• Uses high input of synthetic fertilizers and pesticides	• Uses organic amendments (compost, animal manure, green manure, crop residues) as nutrient sources
• Increases environmental pollution	• Reduces environmental pollution
• Focuses on short-term benefits	• Focuses on long-term productivity
• Degrades soil structure and reduces soil biological diversity	• Improves soil structure and microbial processes by adding organic materials
• Emphasizes less on soil and water conservation	• Emphasizes on soil and water quality management
• Reduces energy use efficiency	• Increases energy efficiency and profit margin
• Emphasizes on quantity of crop products	• Emphasizes on quality of crop products and certification of high quality management
• Generates toxic runoff and pollutes soil and water	• Decreases runoff and soil erosion
• Decreases wildlife habitat and biodiversity (e.g., insects, birds, and beneficial soil organisms)	• Increases biodiversity (e.g. N fixing bacteria)
• Increases risks of food contamination with chemicals	• Produces food free of pesticide, irradiation, herbicide contamination, and other synthetic chemicals
• Uses hybrids and genetically engineered crop varieties to increase crop yields	• Excludes the use of genetically engineered crop varieties
• Reduces C sequestration and increases emissions of greenhouse gases from chemical elaboration and application	• Decreases emissions of greenhouse gases and sequesters C through crop rotations and addition of amendments

7.16.5 Soil Erosion

Organic farming reduces soil erosion over conventional farming when the system maintains a more continuous soil surface cover with cover crops, green manure, and residue mulch. Organic farming systems that use intensive tillage to control weeds without additional conservation practices (e.g., rotations, cover crops) may have, however, equal to or even higher erosion rates than conventional farming. In practice, organic farming involves additions of large amounts of plant and animal manures, which enhances activity and diversity of soil organisms (e.g., earthworms), promotes water infiltration, and decreases soil erodibility (Mader et al., 2002).

Less use of tillage and more reliance on biological techniques to control weeds are key strategies to minimize soil erosion in organic farming versus conventional farming. In the absence or reduced tillage, organic materials provide binding agents (e.g., polysaccharides) and promote development of soil structure by stabilizing and strengthening aggregates. Biologically-bound soil aggregates are less susceptible to disintegration. Earthworms and other soil organisms generate organic substances (e.g. gums, waxes, glue-like substances) which bind primary particles into stable micro- and macro-aggregates. Combination of legumes forages with crops in organic farming is important because soil organisms generally prefer legume-based cover crops as a food source. Lower runoff rates in organic farming result from the higher water infiltration rates enhanced by deep-rooted legume species. Soil erosion rates from soils under organic farming can be 30–140% lower than those from conventional farming (Fig. 7.8).

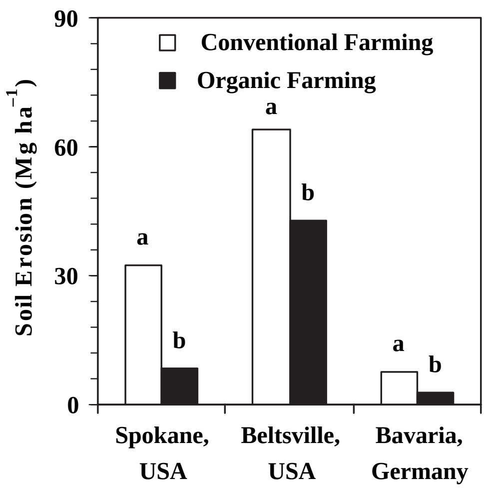

Fig. 7.8 Organic farming reduces soil erosion relative to conventional farming [After Reganold et al. (1987), Green et al. (2005), and Auerswald et al. (2006)]. Bars followed by different lowercase letters are significantly different within each study site ($P < 0.05$). The lower soil erosion rates under organic farming are attributed to better soil granulation, higher macroporosity, and higher water infiltration rates

7.16.6 Soil Biological Properties

The utmost importance of organic farming to soil function is the improvement of soil biological properties. The soil biotic community governs nutrient cycling and availability. Organic farms receiving the same cultural operations as conventional farms normally have higher biological (e.g., earthworm) activity due to the elimination of pesticides (Siegrist et al., 1998). Synthetic chemicals are highly soluble and their excessive use inhibits proliferation and activity of sensitive soil organisms. Surface dwelling earthworms such as *Lumbricus terrestris* are highly susceptible to injury by excessive application of pesticides. Direct contact with pesticides during crawling and feeding on the soil surface can harm earthworms. Dissolved chemicals can percolate through the burrows deep in the soil profile harming even non-surface dwelling earthworms. For example, corn rootworm insecticides and injection of anhydrous ammonia may kill earthworms. Rotation of corn and soybeans with forage legumes generally promotes higher earthworm populations than continuous corn due to elimination of the rootworm insecticides.

7.16.7 Soil Physical Properties

Organic farming enhances aggregation and macroporosity from the addition of biosolids. Soil aggregate stability under organic farming can be 10–60% higher than that under conventional farming systems (Siegrist et al., 1998). Magnitude of soil improvement by organic farming is somewhere in between the no-till and conventional farming systems. The organic farming with intensive tillage may not improve soil properties as compared to no-till, but it often does as compared to conventional farming due to the addition of organic materials (Table 7.4). Frequent and intense tillage in organic farming breaks soil aggregates and accelerates soil organic matter decomposition. Use of more rotations with diverse crops and less tillage operations for weed control reduces bulk density, and increases soil macroporosity, water retention capacity, nutrient supply and cycling, and microbial biomass in long-term organic farming systems.

7.16.8 Crop Yields

Crop yields under organic farming are often lower than those under conventional farming systems. The yield gap between organic farming and other systems depends on management duration, tillage intensity, and source of organic matter in organic farming. A 21-yr study in Switzerland showed that crop yields were 20% lower in organic farming, but the use of chemicals was reduced by 34–53% for fertilizers and by 97% for pesticides, which minimized the differences in net benefits between organic farming and conventional farming (Mader et al., 2002). Agronomic yields under organic farming can decrease by about 30% in crops with high nutrient requirements (e.g., potatoes) and by about 10% in cereal and grasses (Mader et al., 2002).

While crop yields are normally lower, organic farming can be as profitable as conventional farming because of the high market price of organic produce. Besides, the reduced crop yields under organic farming are far compensated by gains in improved soil fertility, reduced energy use, and enhanced biological diversity, and environmental quality. Reduced crop yields in organic farming are common during the transition from conventional farming to organic farming. It takes three

Table 7.4 Organic farming under no-till generally improves soil properties as compared to conventional farming

Soil properties	Management duration (yr)	Conventional farming	Organic farming
[1]Mean weight diameter (mm)	5	1.7a	2.5b
[1]Bulk density ($Mg\,m^{-3}$)		1.5a	1.4b
[2]Depth to argillic horizon (cm)	39	40a	56b
[2]Soil water content ($g\ 100\ g^{-1}$)		9b	15a
[3]Earthworm density (per m^{-2})	14	137b	299a

[1]Hayden (2006); [2]Reganold et al. (1987); [3]Siegrist et al. (1998).

to five yr for the soil to rebuild its natural fertility and stimulate the regrouping of soil organisms following the cessation of conventional farming. The transition period is often called “learning curve” where yields in organic farming lag behind those of the conventional farming. Biological rebuilding of soil fertility is slow but more sustainable once achieved. As the number of soil organisms increases over time more break down of organic materials occurs, increasing nutrient availability to plants. Soil organisms also absorb and retain nutrients in the bodies, reducing risks of nutrient leaching and allowing greater nutrient availability to plants over extended periods of time.

Summary

Conserving soil and water depends on the management of tillage and cropping systems. Well-designed cropping systems enhance soil fertility, reduce soil erosion, and improve soil properties. Diversification of crops promotes biological activity, nutrient cycling, and soil rejuvenation. Management of cropping systems involves management of tillage, crop residues, nutrients, pests, and erosion control practices. Appropriate choice of cropping systems is a strategy to minimize environmental pollution. Crop rotations and organic farming are examples of effective cropping systems for reducing soil erosion and water pollution. The selection and design of cropping practices are a function of soil, management, and climate conditions. Cropping systems include fallow systems, monoculture, strip cropping, multiple cropping, contour strip cropping, crop rotations, cover crops, mixed and relay cropping, and organic farming. Whereas monocropping allows specialization in a specific crop and reduction cultural operations and costs of farm equipment, it reduces crop diversity, deteriorates soil properties, increases the use of fertilizers and pesticides, induces weed and pest invasions, and reduces crop yields.

Crop rotations can consist of single crops, short rotations, and extended rotations. Long rotations are preferred over monocropping and short rotations. In the U.S., Corn Belt region, corn-soybean rotation is the main cropping system although demands for expanded uses of corn (e.g., biofuel feedstocks) may favor monocropping with corn. Rotating row crops with legumes and perennial grass are strategies for managing soil erosion. Dense and permanent vegetative cover not only intercepts the erosive forces of water and wind but also stabilizes the soil underneath. Rotations also improve soil properties by promoting aggregation and macroporosity. Multicropping, which consists of growing more than one crop per year, include double cropping, intercropping, relay cropping, and others. Contour farming and strip cropping are practices that reduce soil erosion in sloping croplands. Organic farming is a system that eliminates the use of synthetic fertilizers, pesticides, and growth regulators to produce food and fiber. It is an ecological approach to improve the soil’s natural fertility and biology.

Organic amendments are used instead of inorganic fertilizers to supply essential nutrients to plants. Organic farming is a promising technology to reduce the excessive use of chemical fertilizers, lower production costs, use the high market prices, and promote environmentally friendly systems.

Study Questions

1. Discuss differences among intercropping, contour cropping, and strip cropping in relation to design and erosion control effectiveness.
2. What would be the impacts of corn monocropping for biofuel production on soil erosion and long-term soil productivity?
3. Suggest the type of crop rotations that would be practiced to provide biofuel feedstocks.
4. Discuss differences in organic farming practiced before and after pre-modern era.
5. Discuss the benefits of organic farming on crop yields and soil properties.
6. Contrast the benefits of corn-soybean rotation against complex and diverse rotations for soil and water conservation.
7. Describe the differences between organic farming and no-till systems.
8. List the soil properties than can be improved by organic farming.
9. Define and discuss the importance of computing the LER.
10. Compare the cropping efficiency of monocropping of corn and soybean with strip cropping with the same crops. Yield of corn was $6\,\mathrm{Mg\,ha^{-1}}$ and that of soybean was $2.3\,\mathrm{Mg\,ha^{-1}}$ when monocropped. Under strip cropping, yield of corn increased to $7.5\,\mathrm{Mg\,ha^{-1}}$ and soybean yield decreased to $2.0\,\mathrm{Mg\,ha^{-1}}$.

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Chapter 8
No-Till Farming

8.1 Seedbed and Soil Tilth

Seedbed refers to "the physical state of the surface soil which affects the germination and emergence of crop seeds," while tilth is "the physical condition of soil as related to its ease of tillage, fitness as a seedbed, and its impedance to seedling emergence and root penetration." (SSSA, 2008). The concept of soil tilth is still evolving. Current definitions of soil tilth are somewhat subjective and qualitative because of the highly dynamic nature and complexity of the soil. Soil tilth is the product of complex interactive processes varying over space and time. In this Chapter, soil tilth is defined as the physical condition of a soil described by its complex and dynamic macro- and micro-scale physical, hydrological, thermal, chemical, and biological attributes affecting tillage, seedling emergence, root penetration, and plant growth.

8.2 Factors Affecting Soil Tilth

Soil tilth is influenced by:

- tillage and cropping systems
- soil attributes and landscape characteristics
- soil management (e.g., residue mulch, manuring)
- soil properties (e.g., texture, clay minerals, faunal activity, organic matter content)
- climate
- time

Tillage directly affects tilth because it loosens and mixes the soil, inducing transient improvements in soil tilth. Tilth changes as the loose soil consolidates with time after tillage. Tilth index varies over the cropping season, increasing with tillage and planting operations and then decreasing with time until harvest (Singh et al., 1992). Conservation tillage, crop residue return, and establishment of cover crops improve soil tilth. Identification of an optimum tillage operation for crop establishment and production is critical. Soil tilth is a qualitative parameter and often based on field experience rather than on a systematic, quantitative, and well-defined approach. It is complex, variable, and site dependent. An accurate assessment of soil tilth is essential to determining an optimum tillage management needed to maximize crop yield.

8.3 Tilth Index

Soil tilth is characterized using a tilth index based on easily measurable soil properties (Fig. 8.1). Tilth index is a quantitative value that describes the soil physical condition ranging from 0 to 1, with 0 being for the worst and 1 the best soil physical condition in relation to crop production (Singh et al., 1992). This index is used in various parts of the world to predict changes in soil productivity and identify the type of tillage needed to achieve an optimal crop production for a particular soil. A well-defined index is important to eliminate the unnecessary extra tillage traffic, thereby reducing costs of production and risks of soil degradation. One of the first simple models developed to estimate tilth index is the following (Singh et al., 1992):

$$Index = CF_1 \times CF_2 \times CF_3 \ldots\ldots\ldots\ldots \times CF_n \tag{8.1}$$

where *CF* is tilth coefficient which varies from 0 to 1, and *n* is number of soil properties needed to evaluate the soil tilth. In this model, the values of each *CF* are computed for each soil property under consideration. The *CF* values are used as multiplicative factors to determine the tilth index. Three defined criteria are used to determine the *CF* for each soil property including non-limiting, critical, and limiting levels of crop growth. For example, a *CF* value of 0 (most limiting factor) is assigned to soil bulk density values $>1.8\,\text{Mg m}^{-3}$ while a value of 1 (least limiting factor) is assigned to densities $<1.3\,\text{Mg m}^{-3}$. The tilth index values are regressed against crop yield data to identify the optimum soil tilth. The number of soil parameters for

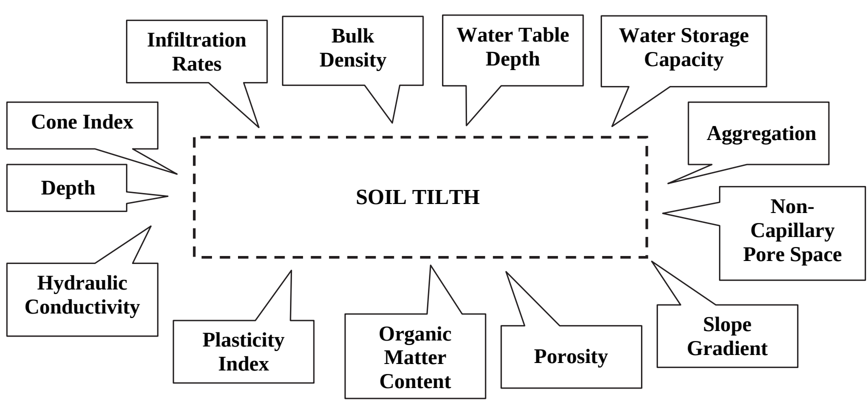

Fig. 8.1 Soil parameters commonly used as input to compute tilth index

computing tilth index varies depending on soil type and tillage system. Plant growth is also a sensitive indicator of soil tilth because it integrates all the plant growth factors including tillage, soil type, cropping system, and climate. Another simple model to evaluate tilth index is shown in Eq. (8.2) (Tripathi et al. 2005):

$$Y = a + b_1X_1 + b_2X_2 + b_3X_3 + \ldots\ldots\ldots\ldots\ldots + b_nX_n \tag{8.2}$$

where Y is crop yield, a, b_1, b_2, and b_3 are regression coefficients, and X_1, X_2, X_3, and X_n are soil properties.

$$A_i = \frac{R_i^2}{\sum_{i=1}^{n} R_i^2} \tag{8.3}$$

$$\text{Index} = \sum_{i=1}^{n} A_i X_i \tag{8.4}$$

where n is number of soil properties. The X_i is normalized by dividing the observed value of the property by its maximum value. The index computed using the normalized values is plotted against the crop yield to obtain an optimum tilth index corresponding to the maximum yield.

8.4 Tillage

Tillage refers to the mechanical operations performed for seedbed preparation and optimum plant growth. A system of tillage involves a sequence of mechanical operations including tilling the soil, chopping and incorporating crop residues, planting crops, controlling weeds before and during plant growth, applying fertilizers and pesticides, and harvesting crops. Tillage is as old as the settled agriculture and has been an important component of traditional farming. It was probably one of the first inventions made by humans in order to grow plants and produce food for survival. Tillage alters the soil tilth and the nature of the whole soil system. Even manual tillage implements can modify soil tilth and cause significant amounts of erosion. Choosing the right tillage tool is crucial for soil and water conservation. While tillage is synonymous to agricultural production, its choice and management are becoming increasingly important to minimizing risks of soil degradation, sustaining agricultural production, and improving soil and environmental quality. The way a soil is tilled determines the amount of residue cover, nutrient availability, soil compaction, soil structural stability, soil-water relationships, soil temperature, and biological activities.

8.5 Tillage Tools

The first tillage implement used in the ancient world (e.g., Babylon) consisted of a wooden device designed to loosen the soil and place seeds (Derpsch, 2001). One of the first common tools called "ard" was pulled first manually and later by animals (Lal et al., 2007). These simple tools merely scratched the surface layer and caused little soil disturbance. The first plows were not designed to overturn the surface layer but for placing seeds and kill some weeds. They were basically scratch-plows and vertical wooden sticks that formed small holes or furrows in the topsoil. The old plows were simpler and smaller than modern tools. Machine-powered plows were invented during the 18^{th} and 19th century. Plows varying in form, shape, and size have been developed over time. The introduction of animal power plows probably around 6th millennium BC, was the start for the increase in size of plows (Lal et al., 2007). The ard evolved into the "Roman plow" around 1 AD and then into a plow that inverted soil.

The sophisticated plow known as "moldboard plow" was used in Europe in the 18th century, and it was specifically designed to overturn the soil and control weeds (Derpsch, 2001). The moldboard plow typically consists of a number of asymmetric and evenly spaced arrow-shaped blades designed to slice the soil horizontally and invert it as the plow moves forward. The first plows turned the soil in one direction only unlike modern plows which are reversible. Various blades or plow-shares are mounted on the runner attached to a tractor. The aggressive plows invert the plow layer and make wide turns unlike manual- or animal-drawn tools. These tractor-pulled devices gradually replaced the manual- or animal-drawn plows. The first moldboard plow in the U.S. was designed in 1784 and marketed in the 1830s by John Deere (Lal et al., 2007). The use of moldboard plow increased rapidly in the 1900s.

8.6 Types of Tillage Systems

Tillage systems are grouped into two main categories: *conventional tillage* and *conservation tillage*:

1. ***Conventional tillage*** is any tillage system that inverts the soil and alters the natural soil structure. It primarily refers to moldboard plowing, which is the ultimate means of soil disturbance.
2. ***Conservation tillage*** is any system that reduces the number of tillage operations, maintains residue cover on the soil surface, and reduces the losses of soil and water relative to conventional tillage. It is a set of innovation technologies including no-till and various reduced or minimum tillage systems such as mulch tillage, strip tillage, and ridge tillage. Reduced or minimum tillage includes any system in which a soil is disturbed less than in conventional tillage but more than in no-till.

8.7 Conventional Tillage: Moldboard Plowing

The moldboard plow was designed to control weeds, loosen compact soils, incorporate residues and fertilizers into the soil, and improve seed germination. Thus, introduction of moldboard plow changed the shape of the fields and increased the size of the cultivated area. It increased food supply particularly in developed countries. As mechanized agriculture spreads, the same sophisticated plow that revolutionized agriculture around the world is being increasingly viewed as responsible for causing soil degradation. Intensive plowing causes soil erosion (e.g., the Dust Bowl), depletes soil nutrients, and reduces biological activities. Because of its adverse impacts, use of moldboard plow has decreased since 1970s, especially in the U.S., Canada, Brazil, Argentina, Australia, etc.

8.7.1 Residues

One of the major factors by which moldboard plowing influences soil productivity is by altering the amount of crop residue left on the soil surface. Moldboard plow chops and buries the residues in the soil. Because plowing leaves little or no residue cover, it increases soils' susceptibility to wind and water erosion (Fig. 8.2). Surface cover, essential to erosion control, is a direct function of tillage intensity. It affects the physical and chemical processes and attributes of the soil. Bare plowed soils are extremely susceptible to crusting and surface sealing. Raindrops striking on bare soil disrupt aggregates and slaking leads to formation of thin films that clog up the pores and reduce water infiltration capacity.

Fig. 8.2 Comparison between a plow tillage (*left*) and no-till (*right*) system on a silt loam (Photo by H. Blanco)

8.7.2 Soil Properties

Excessive tillage increases soil erodibility of the soil, destroys natural soil architecture, reduces microbial processes, and degrades soil tilth. It dramatically reduces aggregate stability and the number of soil organisms such as earthworms, which are important to loosening the soil, recycling nutrients, and creating macropores for increased water-air and gaseous exchange in the soil. The loose soil structure following tillage is highly unstable. Plowed soils are sensitive to internal capillary forces, wetting and drying, and crusting and surface sealing. The greater the surface sealing and thicker the crusts, the lower the water infiltration and higher the runoff rates. Crusts can also reduce seedling emergence, plant growth, and crop yields particularly in fine-textured soils.

8.7.3 Soil Compaction

Plowing reduces soil compaction parameters immediately after tillage early in the growing season, but these improvements are transient and are nullified later in the season (Table 8.1). Large pores created by tillage collapse rapidly because of soil re-compaction, rainfall-induced consolidation, and reduction in soil organic binding agents. The high surface roughness on recently plowed soils increases surface water retention and improves water infiltration. This improvement, however, is short-lived because bare clods are easily detached and eroded by rain and runoff. Tillage reduces soil structural stability and microbial processes.

Table 8.1 Short- and long-term effects of conventional tillage

Transient Benefits	Long-term Consequences
Reduces soil compaction	Increases soil compaction upon rapid soil consolidation
Increases soil porosity	Reduces soil macroporosity and biological activity
Eliminates crusting and surface sealing	Induces severe crusting and seal formation
Accelerates release of essential nutrients upon decomposition of organic matter and increases nutrient uptake	Decreases the soil organic matter content and nutrient cycling and availability
Improves fluxes of water, air, and heat	Decreases hydraulic conductivity and air permeability
Reduces runoff because of increased surface roughness	Decreases infiltration rate and increases runoff
Promotes rapid emergence and plant growth by loosening the soil	Decreases crop production due to reduced water storage and increased evaporation

8.8 Conservation Tillage Systems

Conservation tillage is an alternative to conventional tillage. Any tillage system that leaves at least 30% of residue cover on the soil surface is called conservation tillage (SSSA, 2008). This definition is, however, too narrow to define the appropriate tillage systems that effectively conserve soil and water. While the 30% of residue cover may be appropriate for some soils, it is insufficient in others to reduce soil erosion to permissible levels. When combined with prudent management of crop residues, crop rotations, and cover crops, conservation tillage is a useful technology for protecting soil and increasing/sustaining crop production. The advent of pre-emergent herbicides around 1950s has facilitated the introduction of reduced tillage and other conservation practices. Conservation agriculture occupies about 100 Mha of land worldwide (Derpsch, 2005). Conservation tillage such as the reduced tillage is an evolving system of farming and is not based on standard or fixed tillage systems. The principles of conservation tillage have evolved since 1960s and are becoming widely accepted.

The optimum conservation system should have enough vegetative cover or crop residues to increase soil surface roughness and improve the infiltration capacity, prerequisites for reduction of runoff and soil erosion. A conservation tillage system must be specifically designed for each soil based on site-specific criteria (e.g., farm profitability, severity of soil erosion, soil type, topography, climate).

8.9 No-Till Farming

No-till or zero tillage refers to a system where a crop is planted directly into the soil with no primary or secondary tillage (SSSA, 2008). It is an extreme form of conservation tillage in which soil remains undisturbed at all times except during planting. It is a practice that leaves all surface residues (stalks, cobs, leaves, etc.) on the soil following harvest (Fig. 8.3). Weeds are normally controlled with herbicides unless proper cropping systems such as crop rotations and cover crops are used as supporting conservation practices.

A narrow and shallow furrow is created using coulters or in-row chisels to place the seeds. Theoretically, the term no-till may not be a suitable name if soil is significantly disturbed at planting through the opening of transient furrows for seed placement. The amount of disturbance needed at planting is a function of soil compaction, amount of residue cover, and other site-specific characteristics.

No-till farming represents a new paradigm of soil management for conserving soil and water (Table 8.2). It is part of a technological revolution that is changing the face of agriculture in many regions around the world (e.g., U.S., Brazil, Argentina, Paraguay), and many farmers are switching from conventional tillage (moldboard plow) to no-till. There are marked differences between no-till and the traditional or conventional tillage systems (Table 8.3).

Fig. 8.3 Long-term no-till soil (*left*) next to an intensively moldboard plowed soil (*right*) in a clay loam soil (Photo by H. Blanco)

Table 8.2 New and old paradigms of soil management

Old Approach (Conventional Tillage)	New Approach (No-Till Farming)
• Tillage is indispensable to crop production	• Moldboard plowing is not needed
• The goal is to produce crops	• The goal is to produce crops while conserving soil and water
• Crop residues are either burned or plowed into the soil	• Residues are left on the soil surface
• Soil often remains bare between cropping seasons	• Soil remains covered with residues and/or cover crops at all times
• Crop rotations and cover crops are optional	• Crop rotations and cover crops are part of the management system.
• Risks of soil erosion are high	• Risks of soil erosion are minimum or negligible.
• Pests are controlled with the use of chemicals.	• Biological controls (e.g., crop rotations, cover crops) are used in conjunction with other measures against pests.

8.9.1 Americas

In the U.S., the Dust Bowl in the 1930s stirred the interest in conservation tillage. The Dust Bowl brought changes in agricultural systems and raised questions about the implications of conventional tillage for controlling soil erosion and maintaining crop yields (Phillips, 1973). Mulch tillage was one of the first conservation practices used in the Great Plains to reduce wind erosion. The development of a variety of herbicides, establishment of demonstration sites, and introduction of fluted coulter

Table 8.3 Consequences of conventional tillage and no-till farming (After Derpsch, 2001)

Conventional Tillage	No-Till Farming
• Increases rates of runoff and soil erosion.	• Reduces runoff and soil erosion.
• Degrades soil physical conditions.	• Improves soil physical conditions.
• Increases wind erosion.	• Reduces wind erosion.
• Causes non-point source water pollution with sediments and chemicals.	• Improves surface water quality by reducing losses of suspended and dissolved loads.
• Increases soil organic matter decomposition and emission of greenhouse gases.	• Increases soil organic matter content in the plow layer.
• Depletes soil nutrients.	• Recycles nutrients through residue retention.
• Causes large fluctuations of soil temperature and water content, which negatively affects crop production.	• Reduces fluctuations of temperature (no-till soils are often warmer at night and cooler during daytime during the growing season).
• Causes loses of water by evaporation	• Decreases soil evaporation.
• Reduces soil water retention and plant available water.	• Increases soil water retention and plant available water.
• Reduces water infiltration rates.	• Increases soil macroporosity and water infiltration rate.
• Degrades soil structural properties over time.	• Improves soil structural properties (e.g., aggregate, stability).
• Decreases population and activity of soil organisms.	• Promotes microbial processes (e.g., earthworm population and activity).
• Leads to a gradual reduction in crop yields.	• Sustains crop production.
• Increases costs of production (e.g, labor, time, machines, and fuel).	• Reduces costs of production.

planters enabled farmers to adopt no-till in the early 1960s (Moody et al., 1961). In the U.S., the area under no-till has increased from 5 Mha in the 1980s to about 22 Mha in the 2000 which represents $<18\%$ of the total cultivated area (Lal, 1997a). No-till farming is expected to increase to about 75% of the cropland area by the year 2020 (Lal, 1997a). It is most popular in the U.S. Corn Belt region and Northern plains where problems of water erosion are significant. There is a slow but steady expansion in no-till technology in the USA.

Latin America is experiencing the fastest expansion of no-till farming in the world. No-till farming was introduced in early 1970 in Brazil and Argentina and in late 1970s and 1980s in Paraguay, Bolivia, Mexico, and other Latin American countries. The pioneering no-till trials in Latin America were conducted in Brazil and then in Argentina mainly for producing sorghum and soybean. Unavailability of no-till machines and problems associated with weed control slowed no-till adoption in the 1970s.

Now with the development of cheaper herbicides and an easy access to no-till planters, the technology is expanding throughout Latin America. What started in small research plots has revolutionized the agriculture in the tropics and semi-tropics of Latin America. Area under no-till farming doubled between 1990 and 2000 (Fig. 8.4). In South America, area under no-till has increased from 0.7 Mha in 1987 to 40.6 Mha (60-fold) in 2004 with the largest cultivated areas in Brazil, Argentina, and Paraguay. In Brazil, the area under no-till was only about 0.4 Mha in early 1980s, but now it is >13 Mha (Fig. 8.4). In Paraguay, nearly 70% of the cultivated land is under no-till, representing the country with the largest no-till adoption in the world in terms of percentage of cultivated land. In eastern Bolivia, no-till is also becoming popular for growing sorghum, sunflowers, corn, soybeans, wheat, rice, and even cotton. In Mexico, about half a million ha of land is under no-till, which is also expanding to Central American countries.

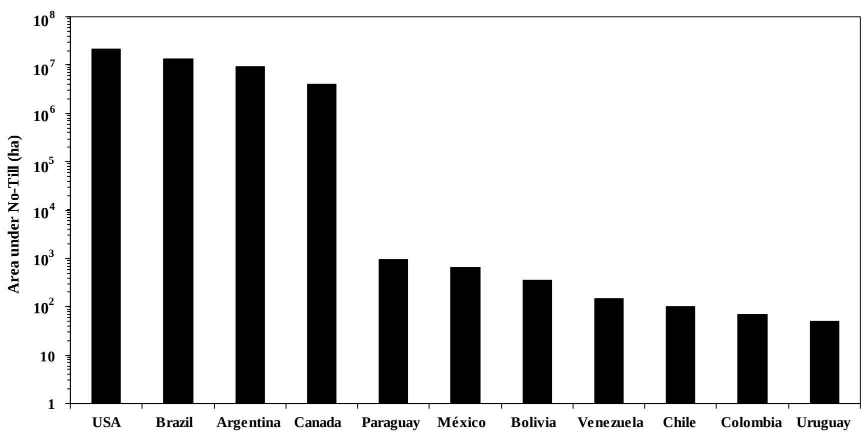

Fig. 8.4 Area under no-till farming in the Americas in 2001 (After Derpsch, 2001)

Soybeans, corn, oats, lentils, sorghum, wheat, barley, sunflower, and beans are the main crops grown with a no-till system. In some regions, no-till systems are being integrated with crop rotations and green manure cover crops. About half a million ha of land are under no-till in irrigated rice paddies in the tropics of South America (e.g., Brazil). Reduced production costs are appealing to farmers although large-scale producers have been more receptive to no-till technology.

8.9.2 *Europe*

No-till farming in Europe started in the 1950s. Abundant residues on the soil surface and restrictions on straw burning induced proliferation of weeds, slowing a rapid expansion of no-till. The lower production costs in machinery, fuel, and labor under no-till are attractive to farmers over conventional tillage because whatever reductions in crop yields under no-till are easily compensated by the reduction in production costs and improvement in soil and environmental quality. Direct planting without plowing saves time and energy. About 16% of the cultivated soils in Europe are highly prone to degradation (Holland, 2004). Despite the many advantages, wide-scale adoption of no-till in Europe, Africa, and Asia is still limited when compared to that in the USA and South America. Field data on benefits of no-till farming are also scarce. About 0.3 Mha of the cultivated area under no-till in Spain and 0.15 Mha in France (Derpsch, 2005).

8.9.3 Africa and Asia

Adoption of no-till technology is also slow in Africa and Asia. Pioneering research work in no-till for Africa started in Nigeria in early 1970s (Lal, 1974; 1976). Studies in African countries including Nigeria, Kenya, South Africa, Tanzania, Zimbabwe, Liberia, and Ghana show that no-till is being used to grow corn, wheat, cotton, and sorghum. Despite the significant research work, adoption of no-till technology is still limited in these regions. In India, area under no-till with wheat has increased from 400 ha in 1998 to about 2.2 Mha in 2005 (Derpsch, 2005). No-till is also being practiced in Malaysia and Sri Lanka. The limited use of no-till in Africa and Asia is probably attributed to various problems associated with (1) high cost of importing no-till equipment for mechanized farms, (2) land tenure, (3) harsh climate conditions, (4) knowledge gap, and (5) lack of crop residue mulch. The area under no-till in South Africa is about 0.3 Mha while that in China is about 0.1 Mha (Derpsch, 2005).

8.9.4 Australia

Australia is one of the places where use of no-till is advancing rapidly, but its adoption varies across regions. The area under no-till is approximately 9 Mha (Derpsch, 2005). About 85% of the cultivated area in western Australia is under no-till and about 40% nationally (ABARE, 2003). In southern Australia, adoption of no-till has been slower. In some regions, development of herbicide resistant weeds is a major constraint to the increase in area under no-till. Introduction of no-till with residue and stubble retention has increased cropping intensity and reduced soil degradation, but the use of herbicides has increased (Radcliffe, 2002). Adoption of no-till is expected to increase steadily as alternative measures for weed control become available.

8.10 Benefits of No-Till Farming

The no-till farming is among the top of the portfolio of strategies to control soil erosion and reduce tillage costs. It is also a unique option to maintain crop productivity and environmental quality. It conserves soil and water while improving soil tilth and increasing soil organic matter. The performance of no-till systems for improving soil functions depends, however, on the soil-specific, topographic, and climate characteristics. The major beneficial impacts of no-till are particularly noted within the upper soil horizons where most crop residues are concentrated. Most of the beneficial aspects of no-till technology are attributed to the crop residues mulch. Thus, no-till systems which leave little or no crop residues after harvest may affect soil properties as adversely as does conventional tillage. Residue left on the surface of no-till soils absorbs and buffers the erosive energy of raindrops and generally improves soil properties (Fig. 8.4; Table 8.4). This buffering process reduces aggregate detachment and surface sealing and crusting, thus decreasing risks of runoff and soil erosion.

Table 8.4 Influence of tillage systems on selected hydraulic properties

Soil	Tillage Duration (yr)	Plow Tillage	No-till
Infiltration Rate ($mm\,h^{-1}$)			
Loam[1]	6	170	120
Silt loam[2]	>15	24	82
Clay loam[3]	16	164	373
Clay[4]	12	83	375
Saturated Hydraulic Conductivity ($mm\,h^{-1}$)			
Silt loam[5]	>15	4	6
Loam[6]	3	37	73
Silt clay loam[6]	3	43	8

[1]Singh and Malhi (2006), [2]Shukla et al. (2003), [3]Singh et al. (1996), [4] De Assis and Lancas (2005), [5] Blanco-Canqui et al. (2004), and [6]Khakural et al. (1992).

8.10.1 Soil Structural Properties

Interactive effects of absence of soil disturbance and residue mulch cover under no-till improve soil aggregation, aggregate stability, macroporosity, soil water retention, water infiltration rate, and hydraulic conductivity when compared to conventionally tilled systems, which break aggregates and reduces soil structural stability. Perhaps, the most sensitive soil parameter to no-till farming is aggregate stability. Aggregates in no-till soils are generally more water stable than in plowed soils because of greater aggregate-binding soil organic matter agents (Fig. 8.5). Increased biological bonding materials from residues enmesh soil particles in clusters, develop water-stable aggregates, and increase the formation of macro-aggregates. Size of aggregates can increase with increase in duration of no-till (Fig. 8.5). Improvement in aggregate stability is important to reducing soil erodibility. The magnitude of improvement in soil structural properties depends on the soil type, management duration, amount of residue return, topography, and climate. The duration of no-till farming is crucial to evaluate its impacts on soil properties. A study conducted in Brazil showed that

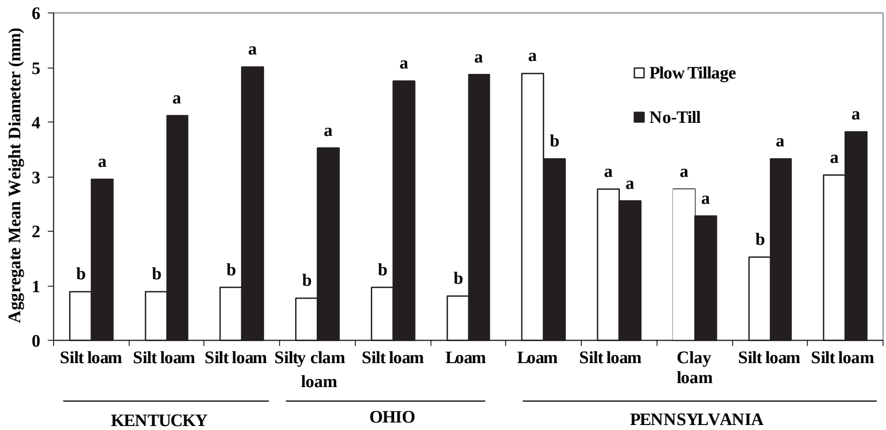

Fig. 8.5 No-till farming impacts aggregate size and stability (After Blanco-Canqui and Lal, 2008). Bars followed by different lowercase letters are significantly different within each soil ($P < 0.05$)

aggregate stability increased whereas bulk density decreased following the adoption of no-till in the first 12 yr (De Assis and Lancas, 2005).

Water infiltration rates and saturated hydraulic conductivity tend to be higher under no-till than in plowed soils because of abundant macropores (Table 8.4). Macropores remain intact in no-till soils. Earthworms can increase water infiltration by 10–100 times depending on the soil (Edwards et al., 1990). Increase in water infiltration rate reduces runoff losses. Residue cover reduces surface sealing of open and continuous macropores, which are major conduits for water flow and gaseous diffusion and transport. Surface residues intercept and retain runoff water and increase the runoff water infiltration opportunity time. Presence of continuous macropores increases the hydraulic conductivity and can offset any reductions in hydraulic conductivity due to compaction.

8.10.2 Soil Water Content

No-till management also impacts soil water storage. Because of abundant residue cover, no-till soils store more water than bare and plowed soils. Residue mulch reduces the evaporation rates, and thus soil water content increases with increase in rates of residue application (Fig. 8.6). Unmulched soils wet and dry quicker than residue-covered no-till soils. No-till farming moderates water balance by reducing runoff, evaporation and excessive percolation.

$$\Delta \text{ water storage} = \text{Input} - \text{Output}$$

$$\begin{aligned}\Delta \text{ water storage} &= \text{Rainfall} + \text{Irrigation} + \text{Capillarity} \\ &\quad - (\text{Evaporation} + \text{Runoff} + \text{Percolation})\end{aligned}$$

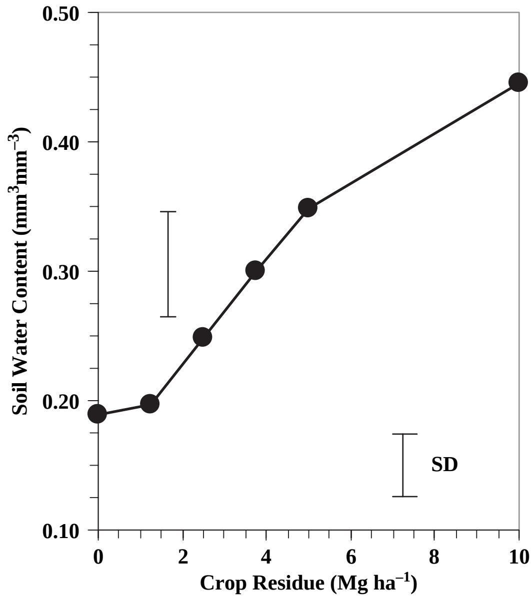

Fig. 8.6 Changes in soil water content due to residue management across three soils under long-term no-till in Ohio (After Blanco-Canqui et al., 2006)

Higher organic matter content enables no-till soil to retain more water than in tilled soils. The magnitude of increases in water retention varies with water potential and residue amount. Because no-till soils tend to have relatively more macropores, differences in soil water retention between tilled and no-till soils at high suctions (more negative) may not be significant. Interaction of organic materials with soil inorganic particles increases plant available water in no-till soils, or the amount of water retained between -0.033 and -1.5 MPa water potentials.

8.10.3 Soil Temperature

No-till management also moderates soil thermal regimes. Moderation of soil temperature is essential to all physical, chemical, and biological processes in the soil. Soil temperature affects seed germination, root and shoot growth, evaporation, soil water storage and movement, microbial processes, nutrient cycling, and many other dynamic processes. Soil temperature also controls C cycling by influencing the temporal and spatial variations of CO_2 fluxes within the soil. Soil temperature is a function of the amount of surface residue cover (Fig. 8.7). Thus, residue removal or addition rapidly alters the soil temperature dynamics essential to soil processes and agricultural productivity. Residue burial in plowed soils reduces the amount of residue left on the soil surface and has negative consequences on soil thermal processes. Residue mulch insulates the soil and buffers the abrupt fluctuations of soil temperature. It moderates the near-surface radiation energy balance and the dynamics of heat exchange between the soil and the atmosphere. Soils without

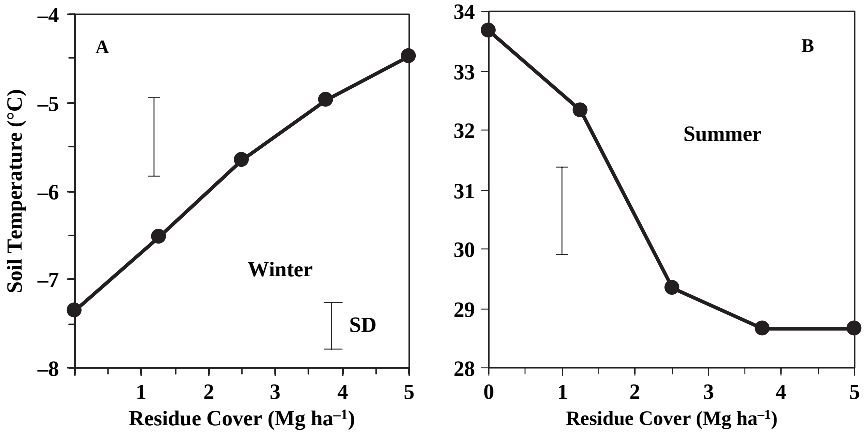

Fig. 8.7 Soil temperature response to changes in residue cover in long-term no-till soils in Ohio

residue mulch are commonly warmer during the day and cooler during the night than residue-mulched soils.

No-till fields create different microclimatic conditions over traditional crop fields. In summer, no-till soils are often cooler than plowed soils, but the opposite is true in winter. Near-surface soil temperature in plowed soils can be 5°C–10°C higher than that in no-till soils in summer time, while it can 2°C–5°C lower than in no-till soils during winter. The increased soil warming in plowed soils accelerates evaporation and reduces available water for crop production during summer. Management of residue in no-till systems is important to control water and heat fluxes for an optimum crop production.

Because no-till soils with residue mulch may be cooler in spring than plowed soils, some producers in cool and temperate regions are reluctant to adopt no-till systems because cooler soil temperatures reduce seed germination and delay plant establishment and growth (Arshad and Azooz, 2003) (Fig. 8.8). Thus, partial removal of residue mulch may be an option to reduce the presumed excessive cooling of some no-till soils during spring.

8.10.4 Micro-Scale Soil Properties

Tillage management influences properties of both whole soil and aggregates (Fig. 8.9). Soil aggregates are the structural elements that influence the behavior of the whole soil. Strength of small aggregates affects soil erosion through its influence on soil detachment, slaking, and water infiltration. Aggregate physical properties influence root growth and seedling emergence, soil water retention, and air flow. Structural aggregates differ in their properties from the bulk soil because these units are characterized by higher internal friction forces and more contact points than bulk soil. For example, the bulk density of discrete aggregate is commonly higher

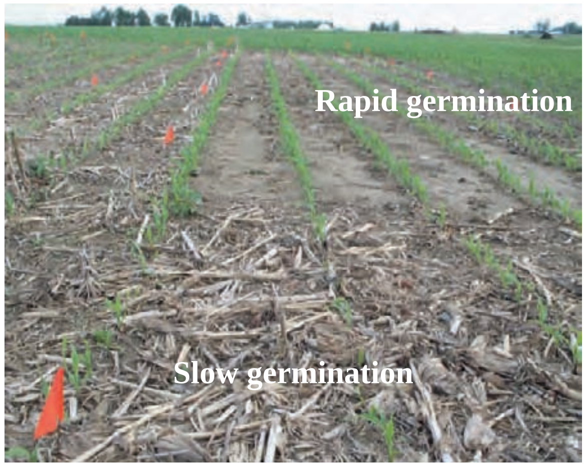

Fig. 8.8 Cool soils under heavy residue mulch slow germination and emergence of corn in no-till systems (Photo by H. Blanco). Soil temperature dynamics in no-till soils under different climatic conditions and seasons must be understood to properly manage crop residues for conserving soil and water. Wet, cool, and clayey soils are the most adversely affected by heavy residue mulching

than that of bulk soil. Aggregates possess an array of strength levels affecting soil compaction.

Aggregates are sensitive to tillage and cropping management systems. Long-term no-till practices impact aggregate strength, density, and water retention capacity different from conventional tillage. Excessive tillage, rapid post-tillage consolidation, and low organic matter concentration in plowed soils alter aggregate formation and properties. Increases in soil organic matter can increase or decrease the strength of aggregates depending on the soil texture, nature of organic matter, and soil water

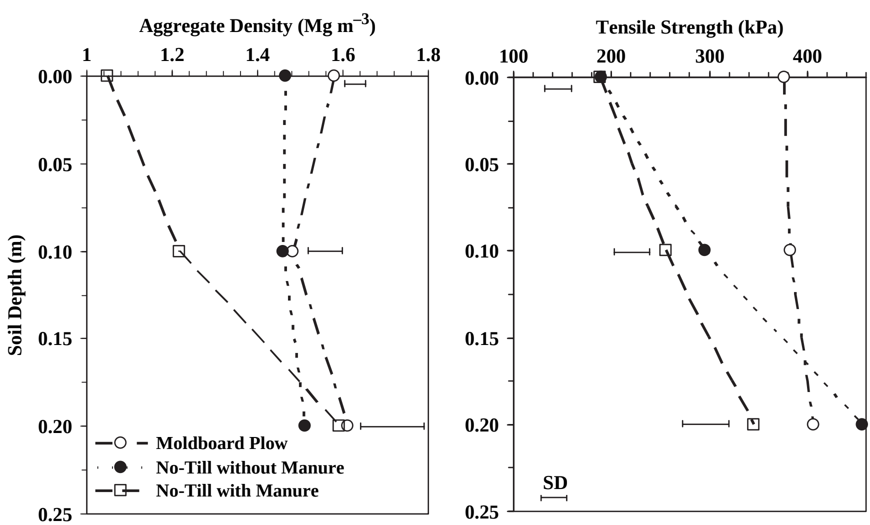

Fig. 8.9 No-till impacts on soil aggregate density and strength in a sloping silt loam (After Blanco-Canqui et al., 2005)

content. No-till management enhances formation of C-enriched macro- and micro-aggregates. Plowed soils often have denser, more compact, and stronger aggregates compared to no-till following post-tillage consolidation (Fig. 8.9). The strength of aggregates tends to increase with increase in no-till -induced changes in organic matter concentrations in clay soils and decrease in silt loam and sandy soils (Imhoff et al., 2002; Blanco-Canqui et al., 2005).

8.10.5 Soil Biota

Permanent residue presence on or near the soil surface is vital to activity and population of soil biota. Soil biota, including macro- and micro-organisms, influence soil aggregation and formation of pores essential to soil structural development and water movement. No-till soils increase earthworm population over tilled systems regardless of cropping system (Fig. 8.10). Earthworm burrowing and decomposition of organic matter are essential processes of aggregation and aeration. No-till systems revitalize the soil and enhance formation and preservation of earthworm macropores.

8.10.6 Soil Erosion

No-till is the most important conservation system because it produces the least amount of soil erosion. It provides a dual function in soil erosion control because it reduces both the effectiveness of *erosivity of r*ain and *erodibility of the soil*. This combined action decreases soil erosion risks compared with practices that bury crop residues (e.g., plow till). Because of the residue mulch cover, no-till reduces the

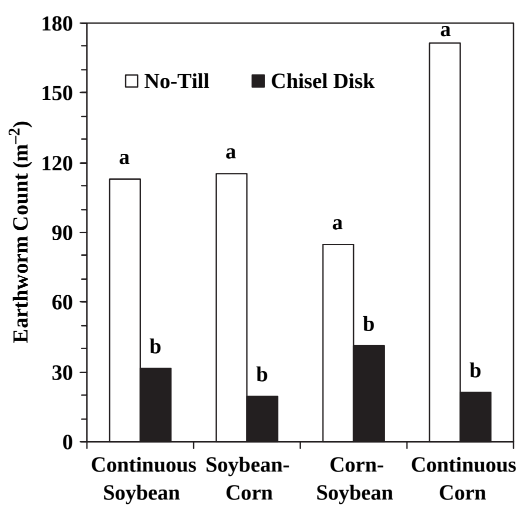

Fig. 8.10 Tillage influence earthworm population (After Jordan et al. 1997). Bars followed with different letters within the same cropping system are significantly different ($P < 0.05$). Residue left on the soil surface provides an abundant food source and habitat to earthworms responsible for macropore network development. Reduction in surface mulch cover reduces earthworm populations and the number of surface-connected macropores

effect of rain erosivity by buffering the erosive energy of raindrops and preventing the direct impact of them on the soil surface. The reduction of raindrop impact decreases aggregate detachment and slaking. Residue mulch also reduces the erosivity of upstream runoff by increasing roughness of the soil surface. The rough surface increases infiltration, reduces the velocity and volume of runoff, and traps eroded sediments.

Runoff and soil erosion decrease with the increase in organic matter content in no-till soils (Rhoton et al., 2002). Soil erosion from no-till soils can be as low as 10% of that from plowed soils (Table 8.5). No-till practice reduces soil erosion by preventing formation of rills. Some erosion can still occur in no-till systems but it takes mostly in the form of interrill erosion. No-till management is more effective at reducing soil erosion than runoff water loss. Runoff leaving no-till fields is, however, less turbid than that from plowed soils because sediment particles in suspension are filtered by residue mulch.

Table 8.5 Differences in runoff and soil erosion from plowed and no-till soils

Soil	Tillage Duration (yr)	Runoff (mm)		Soil Erosion (Mg ha^{-1})	
		Plow Till	No-till	Plow Till	No-till
Sandy loam[1]	4	70	21	7	0.5
Loam[2]	13	15	9	4	2.2
Silt loam[3]	34	29	0.0	3	0.0
Clay loam[4]	3	38	55	1	0.4
Clay[5]	2	61	45	13	1.5

[1]Lal, 1997b, [2]Mickelson et al. (2001), [3]Rhoton et al. (2002), [4]Gaynor and Findlay, 1995, and [5]Cogo et al. (2003).

8.11 Challenges in No-Till Management

There are constraints to the adoption of no-till technology. No-till technology may not always be easily adopted in all soils or regions. Its expansion has been slow due to local and regional soil and climate differences. Performance of no-till farming depends on soil type, climate, and management.

Some of the site-specific challenges with no-till management include:

- Increased risks of soil compaction
- Stratification of soil organic matter, and accumulation in the surface layer
- Increased development of herbicide resistant weeds
- Increased use of herbicides
- Reduced seedling germination due to slow soil warming
- Increased use of N fertilizers
- Increased chemical leaching
- Reduced crop yields

8.11.1 Soil Compaction

Soil compaction may increase with the conversion of till into no-till systems from the lack of transient soil loosening by tillage operations. Field studies have shown that no-till farming impacts on soil compaction are site specific (Blanco-Canqui and Lal, 2007) (Fig. 8.11). Soil compaction under no-till may be particularly considerable in poorly drained clay soils. No-till systems may require some occasional plowing to reduce compaction of the surface soil. Soil compaction is often lower in plowed than in no-till soils immediately after tillage (Fig. 8.11). Soil consolidation after tillage can rapidly compact plowed soils to levels equal to or even higher than that in no-till. Site-specific characterization of no-till performance for an extended period (>10 yr) is desirable to assess magnitude of soil compaction.

Soil compaction normally increases following conversion of till to no-till systems during the early years, but it often decreases as the soil recovers within 3–5 yr after conversion. The recovery is due to the gradual build-up in earthworm population and development of soil structure. Well-structured no-till soils increase continuity and connectivity of biological macropore within the soil profile. Moderate soil compaction may benefit crop establishment because of better root-soil contact particularly in dry years. While compaction may increase in some no-till soils, improvements in other soil properties such as macroporosity, water infiltration, and aeration normally offset the problems of compaction. It also reduces rapid changes in freezing and thawing, and shrinking and swelling, which influence soil compaction.

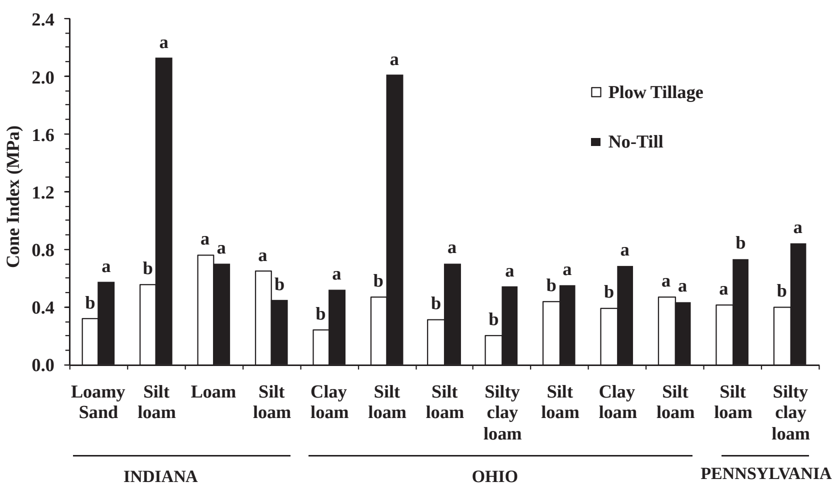

Fig. 8.11 Soil compaction in no-till soils across three states in the eastern USA (After Blanco-Canqui and Lal, 2007). Bars followed by the same letter within the same soil are not significantly different ($P < 0.05$)

8.11.2 Crop Yields

Crop yields from no-till systems may be higher, lower or equal to when compared to those from conventional tillage. No-till does not increase crop yields in all soils. Indeed, crop yields from no-till systems can be lower 5–10% compared to plowed soils with poor drainage and high clay content (Lal and Ahmadi, 2000). The slow soil warming in spring due to high residue mulch cover may negatively affect plant emergence and reduce plant growth in no-till soils. Proper residue management and adoption of other conservation practices such as crop rotations, and cover crops are key to revitalize soil fertility and ensure the success of no-till farming. Economic costs for growing cover crops and introducing crop rotations in monoculture farms can be, however, high in some regions. Markets for products from rotations may not exist. Concerns over decreased crop yields have partly contributed to the slow adoption of no-till in spite of reduced production costs (e.g., labor, fuel, machinery). The overall lower production costs under no-till can nevertheless offset the lower crop yields observed in some soils.

8.11.3 Chemical Leaching

No-till management can have positive or negative effects on nutrient leaching. Some no-till soils may require higher rates of N application to increase crop yields because of reduced mineralization. The higher N application increases concerns over non-point source pollution of water. The proportion of rainfall entering the no-till soil is generally greater than that in tilled soils due to the presence of water-conducting macropores (Shipitalo and Gibbs, 2000). No-till systems create continuous and vertical macropores (e.g. earthworm burrows) often extending throughout the soil profile (Butt et al., 1999), which can increase the potential for preferential flow or bypass flow of water carrying soluble chemicals and causing pollution of downstream waters. Nitrates, for example, can be leached out of the no-till root zone much quicker than from tilled soils. Excessive nitrate leaching in no-till may require higher N fertilization rates to compensate for N losses by leaching for an optimum crop production.

The greater by-pass flow in no-till systems may not, however, carry large amounts of soluble nutrients. The nutrient concentrations in the by-pass water flow can be lower than that in soil matrix water flow because of the limited interaction with soil matrix. No-till system can also decrease nutrient leaching through the use of cover crops and crop rotations, proper timing of fertilizer application, and use of an integrated nutrient management.

8.12 No-Till and Subsoiling

Subsoiling, also known as deep tillage, is a practice that loosens soil to below the Ap horizon without inverting and mixing the plow layer. It fractures and slightly lifts the soil while minimizing vertical mixing. This practice is used to break up compacted subsurface layers that form between 25 and 40 cm below the soil surface from natural consolidation or machinery traffic. This compacted layer, also called plowpan, restricts seedling emergence, root growth, and down- and up-ward water and air movement. In some cases, the soil may be saturated with water above the plowpan and unsaturated below due to the virtual impermeability of the plowpan. Plant roots often concentrate above the plowpans with reduced access to subsurface available water and often wilt when supply of surface water is limited.

Subsoiling can alleviate the above problems. It has been used as a companion practice to no-till. Subsoiling moldboard plowed soil is sometimes desirable before converting the system into no-till. The water content of the compacted layer must be below the field capacity prior to subsoiling. Soils that are too wet during subsoiling can create additional compaction problems. Subsoiling does not always increase crop yields, depending on soil type. Silty clay loam soils appear to respond to subsoiling better than heavy clayey soils. Choice of subsoiling equipment is critical. While subsoiling is designed to allow the practice of no-till in soils susceptible to compaction, some subsoiling machines tend to mix and disturb the whole plow layer. Machines equipped with narrow shanks reduce disturbance of the plow layer and maintain residue cover on the soil surface must be used. Because subsoiling of deep layers can be expensive, controlled traffic decreases the need of subsoiling and prolongs the benefits of no-till farming. Depending on the traffic and soil susceptibility to compaction, subsoiling is done every 3–4 yr.

8.13 Reduced Tillage

Reduced tillage refers to any conservation system that minimizes the total number of tillage primary and secondary operations for seed planting from that normally used on field under conventional tillage (SSSA, 2008). It is also called minimum tillage because it reduces the use of tillage to minimum enough to meet the requirements of crop growth. Reduced tillage is a conservation management strategy that leaves at least 30% residue cover to minimize runoff and soil erosion, improve soil functions, and sustain crop production. Reduced tillage is becoming an important conservation practice like no-till. These systems reduce runoff and soil erosion and improve or maintain crop yields compared to conventional systems. Runoff and soil erosion from minimum or reduced tillage are generally between those from conventional tillage and no-till (Table 8.6). Some of the systems within reduced tillage include mulch till, ridge-till, and strip-till.

8.14 Mulch Tillage

Mulch tillage is a practice where at least 30% of the soil surface remains covered with crop residues after tillage. Tillage under this system is performed in a way that

Table 8.6 Runoff and soil erosion under minimum tillage as compared to conventional tillage and no-till across various soils with different slopes

Soil	Runoff (mm)			Soil Erosion (Mg ha^{-1})		
	Plow Till	Reduced Till	No-Till	Plow Till	Reduced Till	No-Till
Clayey[1]	91	12	7	14.6	1.1	0.6
Clay loam[2]	38	76	55	0.9	0.5	0.4
Loam[3]	12	8	5	1.5	1.4	1.1
Sandy loam[3]	20	10	5	3.0	2.6	1.3

[1]Beutler et al. (2003), [2]Gaynor and Findlay (1995), and [3]Packer et al. (1992).

leaves or maintains crop residues on the soil surface. Mulch tillage is an extension of reduced tillage and is also called mulch farming or stubble mulch tillage. The soil under mulch tillage is often tilled with chisel and disk plows instead of moldboard plows, and thus it minimizes soil inversion.

One of the advantages of mulch tillage over no-till is that it can control weeds better by tillage. Minimizing the secondary tillage is important in mulch tillage to conserve and maintain residue cover. While soil erosion in mulch tillage is commonly lower compared to that in conventional tillage, it can be higher than that in no-till systems because mulch tillage leaves less residue cover on the soil surface than no-till. The use of mulch tillage requires the modification of tillage implements and operations. The choice of implement for mulch tillage is specific to each soil and management.

In the USA, mulch tillage started in the 1930s following the severe droughts and wind erosion of the Dust Bowl. Mulch tillage became popular in the Great Plains over clean or conventional tillage to conserve soil and water. It is best suited for semiarid or drylands because it reduces evaporation and increases plant available water. Mulch tillage can be as effective as no-till systems for conserving soil and maintaining crop yields in drylands. In humid regions and clayey soils, it may not substantially improve soil conditions.

8.15 Strip Tillage

This system is also called partial-width tillage and consists of performing tillage in isolated bands while leaving undisturbed strips throughout the field. By doing so, strip tillage combines the benefits of no-till and tillage. Only the strips that will be used as seedbeds are tilled. The strips between the tilled rows are left under no-till with under residue cover. Strip tillage loosens the tilled strip and temporarily improves drainage and reduces soil compaction. The strip tillage can be an alternative to no-till farming in poorly drained and clayey soils. Where no-till has not maintained or improved corn production, strip tillage is a recommended option. The benefits of strip tillage are many (Vyn and Raimbault, 1992):

- It promotes residue and organic matter accumulation and improves biological activity (e.g. earthworm population). While tillage along the seedbeds alters earthworm dwellings and accelerates residue decomposition, the undisturbed strips can harbor earthworms and accumulate organic matter.
- Tilling in localized strips eliminates excessive mulch cover and speed up soil warming in spring during crop establishment unlike no-till systems.
- Fertilizer is applied primarily to the narrow strips, reducing fertilization rates and application costs.

The strip tillage requires appropriate equipment for reduced tillage. It is often performed using a compact assembly of row cleaner, coulter, shank, and disks with a width equal to that of the planter. The cost of equipment for strip tillage can exceed that for no-till. The row cleaner removes residues from the rows while the coulter and shank break up and loosen the soil to a 10- to 20-cm depth. The disk covers intercept the soil during tillage and keep it from spreading to untilled strips. In continuous corn systems, residues are chopped to facilitate tillage.

8.16 Ridge Tillage

Ridge tillage is a system in which 15- to 20-cm high permanent ridges are formed by tillage during the second cultivation or after harvest in preparation for the following year's crop (Fig. 8.12). The ridges are maintained and annually re-formed for growing crops. Crops are planted on the ridge tops, a practice known as ridge planting. This system is designed to reduce costs of tillage, improve crop yields, and reduce losses of runoff and soil. Ridge tillage can reduce soil erosion by as much as 50% as compared to conventional tillage (Gaynor and Findlay, 1995). A specialized equipment assembly of a ridge-till cultivator, coulter, and disk hiller is used to cut through the residues and form ridges. The disk hiller throws the soil towards the row and forms peak ridges. Shallow scalping (2–5 cm deep) of the ridge tops and residue removal by a row-cleaner are necessary for placing seeds. The residue removal temporarily leaves the ridge crests bare but the residue is moved back during ridge reforming. Also, residue produced at harvest is left on the soil surface to protect the ridge tops. In soils with low ridges, direct planting (no-till) may be preferred over scalping. The ridge tillage is advantageous because:

- Traffic is confined to the rows between ridges. The controlled traffic reduces compaction of the whole field and allows soil structure development within untrafficked ridges.
- Ridges built on the contour create mini-terraces, which serve as permanent structures for soil erosion reduction.
- The residue accumulation in the furrows or depressions slows runoff velocity and reduces soil detachment and transport. The soil removed or lost from the ridge shoulders ends up in the furrows and is protected by the residue cover.

Fig. 8.12 A ridge tillage field used for soil and water conservation (Courtesy USDA-NRCS). Each ridge supports one single row of plants. Tillage and planting are done on the same ridges year to year

- Well-managed ridges concentrate about 30–50% of the original residue on the ridge tops and shoulders.
- Ridges can be 2°C–5°C degrees warmer in spring during planting. The warm soils hasten seed germination and allow early planting of crops.
- Ridges create dry zones and improve soil conditions for growing crops in wet and cool environments. This tillage system is particularly suited to poorly drained and clayey soils where no-till systems and other high-residue tillage systems may fail.
- Costs of ridge tillage are lower compared to conventional tillage.
- Ridge tillage can reduce use of herbicides by about 50% over conventional tillage and no-till. Weeds are controlled by banding herbicide on the ridges only and by ridging operations.
- Crops yields in ridge tillage can be higher than in no-till systems.

Some of the disadvantages of ridge tillage include:

1. Increased tillage costs as compared to no-till.
2. Specific tillage equipment for forming and maintaining ridges and planting crops. The equipment must have the right wheel spacing.
3. If the soil is nearly level or level, ridge tillage may create drainage problems due to water ponding in the furrows.
4. Planting on curved ridges on the contours and sloping soils (>4%) may be difficult.

5. Maintaining ridges at harvesting and planting can be expensive and labor intensive.
6. Runoff and soil erosion can be higher in ridge tillage as compared to no-till.

Summary

Intensive tillage disturbs and mixes the soil, alters soil tilth, and causes soil degradation. Conservation tillage such as reduced tillage and no-till management, in turn, improves soil tilth. Plowing is as old as agriculture itself. The old plows were manual and simple until the introduction of moldboard plow that revolutionized agriculture and increased concerns of soil erosion. Tillage systems are grouped into two main categories: conventional tillage and conservation tillage. The former refers to practices that invert and mix the soil whereas the later refers to practices that reduces or eliminates soil disturbance and leaves most of the residue on the soil surface. Moldboard plowing is the typical practice of conventional tillage. Moldboard plowing breaks up the soil, provides temporary control of compaction and weeds, but it plows all the residues under. Soils without residue mulch are susceptible to erosion and deterioration. Aggressive plowing destroys the natural soil structure and reduces earthworm population and organic matter storage while increasing soil erodibility. Runoff and soil erosion rates are generally greater from plowed than no-till soils.

Conservation tillage includes no-till, mulch tillage, strip tillage, and ridge tillage. No-till is one of the top soil conservation technologies that has changed the way farming and crop residue management is done. No-till combined with complex and diverse crop rotations and cover crops is a strategy for reducing soil erosion. It is an evolving system and its performance depends on site-specific conditions (e.g., soil type, topography, climate). This technology is rapidly expanding in South America, North America, and Australia, whereas its adoption in other regions has been slow due to economic and management constraints. No-till may increase soil compaction or lower crop yields. Thus, occasional subsoiling and tillage may be necessary to ameliorate excessive compaction in no-till systems. Reduced tillage, mulch tillage, strip tillage, and ridge tillage are alternatives practices to no-till for conditions where no-till performs poorly.

Study Questions

1. Define soil tilth and its parameters of evaluation.
2. Discuss the differences that exist among the conservation tillage systems.
3. Is there any difference between no-till and zero tillage?
4. Describe the mechanisms for runoff reduction under no-till systems.
5. Discuss the strategies to ameliorate soil compaction in no-till systems in clayey soils.

6. Describe the worldwide distribution of no-till technology, and factors affecting it?
7. Discuss the constraints for the limited adoption of no-till in developing countries.
8. State the research needs for enhancing no-till adoption.
9. What are the implications of no-till technology for non-point source pollution.?
10. What are the impacts of no-till farming on crop yields as compared to plow tillage?

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Chapter 9
Buffer Strips

Buffers are strips or corridors of permanent vegetation used to reduce water and wind erosion (Fig. 9.1). These conservation buffers are designed to reduce water runoff and wind velocity, filter sediment, and remove sediment-borne chemicals (e.g., nutrients, pesticides) leaving upland ecosystems. Buffer systems are commonly established between agricultural lands and water bodies (e.g., streams, rivers, lakes). When placed perpendicular to the direction of water and wind flow, buffers are effective measures for reducing sediment fluxes. Buffers are a unique ecosystem established between two contrasting systems: terrestrial and aquatic. Their functionality is thus influenced by the interactive effects of both upslope and downslope environments.

Fig. 9.1 Tall fescue filter strip established between a waterway and cropland (Courtesy USDA-NRCS). Buffers are ecotones of the adjoining terrestrial and aquatic landscapes as they integrate fluxes of energy, matter, and living species

The use of buffers is common in many parts of the world particularly in sloping lands and developing regions where access to heavy equipment and construction of mechanical structures (e.g., terraces) can be unachievable for small land holders. In the USA, since 1980's, there has been a great deal of interest in incorporating

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buffer strips into agricultural systems to mitigate environmental pollution. Presently, buffers are among the best management practices for water quality management and their establishment is strongly promoted by initiatives such as the CRP. Indeed, ambitious goals for expansion of buffer strips have been set by USDA-National Conservation Buffer Initiative to install several millions kilometers of buffers across the U.S. croplands. Despite the increased support, adoption of buffer strips is still slow due, in part, to management and economic constraints.

9.1 Importance

Buffers provide numerous and positive benefits to water quality, agricultural production, wildlife habitat, and landscape aesthetics. Buffer strips improve the quality of soil, water, and air (Table 9.1). Buffers can trap $>70\%$ of sediments and $>50\%$ of nutrients depending on the plant species, management, and climate. They are multifunctional systems. Above the surface, buffers reduce the runoff velocity and trap sediments and nutrients. Below the surface, they stabilize the soil in place, bind the soil aggregates, improve the structural characteristics, and increase soil organic matter content and water transmission characteristics. On sloping soils, buffer strips prevent slope failure (e.g., mass movement, land slide) while reducing soil erosion. Buffer strips anchor the shorelines of the water bodies and dissipate the erosive energy of water waves. Buffers are also important to wildlife habitat recovery and protection of biodiversity. They protect livestock and wildlife from wind and snow hazards and provide food, shelter, and safe corridors for wildlife animals and birds.

Buffers have important implications for both rural and urban landscapes. Runoff volume, rate, and peak rate and sediment load increase linearly with increase in urban areas. Increasing urbanization across the globe modifies the character and integrity of the landscape geomorphology and affects the quality of water streams. Concentrated runoff from urbanized areas often creates channelized flow, increasing the runoff capacity to transport non-point source pollutants. Peak runoff flows drastically reduce the effectiveness of structural drainage systems. Thus, well-designed buffer systems can be an important companion to erosion control practices (Fig. 9.2).

Table 9.1 Functions of buffer strips in soil and water conservation

Soil stabilization	Erosion and pollution control	Soil properties	Wildlife habitat
• Anchor the soil in place • Intercept concentrated flow • Stabilize the shorelines or streambanks	• Reduce runoff velocity • Filter sediment and nutrients • Filter pollutants from air	• Increase water infiltration • Increase soil organic matter content • Improve soil structure	• Provide food source for fauna • Provide nest and shelter on habitat for biodiversity • Enhance species biodiversity

Fig. 9.2 Buffers reduce water (*left*) and wind erosion (*right*) and improve landscape aesthetics (Courtesy USDA-NRCS)

9.2 Mechanisms of Pollutant Removal

Understanding of the mechanisms of buffer strips for runoff and sediment control is essential to designing and managing buffers. Buffer strips control sediment and nutrient losses through the following principal mechanisms:

1. ***Decrease of runoff velocity***. Dense and tall vegetation slows the runoff velocity and spreads the incoming runoff above the buffers. Living plant materials and soil surface residues within buffers slow runoff flow, filter sediment, and cause sedimentation. Plant residues at the soil surface or bed sponge up and trap sediments and plant nutrients.
2. ***Stabilization of soil matrix***. Mixture of coarse, medium, and fine plant roots enmeshes, binds the soil particles, and stabilizes the soil matrix.
3. ***Reduction of runoff amount***. Plant roots create a network of channels or macropores through which runoff water can infiltrate into soil, thereby reducing the total amount of surface runoff.
4. ***Runoff ponding***. Depending on the type of vegetative strips (e.g., grass barriers), ponding of sediment-laden runoff on the upstream side of buffers is one of the main mechanisms for sediment deposition and trapping. Reduction in runoff velocity and ponding is correlated with the roughness of buffer strips.
5. ***Water infiltration***. Runoff ponding and delay promote water infiltration along with flocculation of clay or colloidal soil particles.

The mechanisms of chemical removal differ from those of sediment removal and depend on the type and form of chemicals. Sediment-bound organic compounds such as organic N and particulate P are trapped with sediment. Soluble chemicals, in turn, are primarily removed by infiltrating water and absorption by plants and soil microorganisms. Transformation (e.g., denitrification) of chemicals during runoff ponding and slow filtration are also effective means for nutrient removal. Buffers strips are more effective in trapping sediment than plant nutrients because of differences in solubility. Soluble nutrients are mixed with runoff water and are not as easily filtered as are sand and silt particles.

9.3 Factors Influencing the Performance of Buffer Strips

The primary factors affecting the effectiveness of buffer strips are:

1. ***Runoff velocity and rate***. Transport of sediments is a function of velocity and rate of runoff. The greater the velocity and rate of runoff, the greater the sediment transport capacity. Velocity and rate of runoff vary within the same field and affect the runoff transport capacity.
2. ***Flow channelization***. Runoff flow through the buffers hardly follows uniform pathways. It converges and diverges and tends to concentrate in small channels randomly distributed through grass strips. High-resolution topographic surveys and dye tracer studies have shown that runoff flow meanders as it travels through the tortuous grass strips. These dynamic processes of flow channelization and changes in flow depth within buffers reduce the sediment trapping. Channelization increases flow rates and reduces the sediment trapping efficiency.
3. ***Vegetation type***. Dense, tall, and deep-rooted vegetation with stiff stems offers higher resistance to runoff. The effectiveness of filter strips increases with increase in height unless the vegetation (e.g., grass) is overtopped by runoff and sediment load. Tall vegetation with flexible stems is prone to failure.
4. ***Width of strips***. The wider is the filter strips, the greater is the amount of sediment and nutrient trapped. Sediment mass often decreases exponentially with increase in width of tall fescue filter strips. The filter strips retain sediment more when the vegetation height and width interact compared to either increase in height or width of strip alone.
5. ***Soil particle size***. Sand particles and aggregates are more easily trapped than clay particles. Soils with stable aggregate are less dispersed by runoff, generating less sediment than those with unstable or weak aggregates.
6. ***Soil structural characteristics***. Porous soils with high saturated hydraulic conductivity and infiltration capacity reduce runoff and illuviate clay particles and soluble nutrients.
7. ***Soil slope***. Effectiveness of grass strips for reducing pollutants decreases with increase in slope degree and length. Runoff flows faster on steep slopes than on gentle slopes. Transport capacity of runoff significantly increases with increase in slope gradient. Wider (>10 m) buffer strips are required in steeper slopes for reducing the same amount of sediment as in gentle slopes.

8. ***Upland management***. Buffers perform better when combined with other upstream conservation practices. Residue management and use of cover crops improve the effectiveness of buffers for reducing transport of pollutants.
9. ***Size of sediment source area***. The larger is the sediment source, the greater are the runoff volume and sediment load.

9.4 Types and Management

There are a wide range of buffer strips:

1. Riparian buffers
2. Filter strips
3. Grass barriers
4. Grassed waterways
5. Field borders
6. Windbreaks

The design, management, vegetation type, and length and width of strips vary among different types of buffer strips (Fig. 9.3). Trees, shrubs, and native and introduced grass species are used as buffers. Dense vegetation with extensive and deep rooting systems is recommended for buffer strips. A combination or mixture of diverse species such as trees, shrubs, and grasses is preferred over single species for enhanced performance of buffers. Woody buffers are important to stabilizing streambanks while herbaceous buffers with fine roots improve water infiltration and soil structural stability. Sediment and nutrients losses decrease linearly with an increase in root biomass. Trees with large roots also improve drainage by loosening

Fig. 9.3 Grassed buffer strips on the contour integrated with field crops (Courtesy USDA-NRCS)

the soil and by transpiring water. To reduce wind erosion, buffers strips must be dense, tall, and provide continuous surface cover.

Design and management of the systems vary in response to local and regional conditions and needs. Climate, topography, soil type, land use and management influence the selection of species as well as the effectiveness of buffer systems. Buffer strips must be designed based on the wind velocity, anticipated runoff flow depth, and frequency of runoff events. For example, concentrated flow of runoff has more energy and velocity than the shallow interrill flow, thus requiring specific buffer strip designs for its control. Greater flow depths and higher velocities cause more sediment transport and less deposition. In the following sections, different types of buffer strips are discussed. The use and attributes of windbreaks are discussed in Chapter 3.

9.5 Riparian Buffer Strips

Riparian buffer strips are wide strips of permanent mixture of woody and herbaceous vegetation planted along agricultural fields designed to mitigate sediment and nutrient flow to streams (USDA-NRCS, 1999) (Table 9.2). Establishment of riparian buffers is common in the USA and parts of Europe. Riparian buffers are used in both agricultural and urban soils along streams to control sediment transport (Fig. 9.4). Riparian buffers are more widely used than other buffer strips for reducing sediment loads and protecting the aquatic ecosystems from contamination and eutrophication. These buffers consist of grasses, trees, shrubs or a combination of these vegetations. Wide riparian buffers (>10 m) comprised of native plant species filter sediments and benefit the wildlife habitat. Riparian buffers between 5- and 30-m wide can reduce runoff and nutrient export by $>30\%$ (Sheridan et al., 1999).

The effectiveness of woody buffers for sediment reduction is mostly because of improved infiltration rates rather than through soil particle settling as sparse woody trees may not significantly filter sediment from runoff. Riparian buffers may fail to reduce N and P export under large amounts of runoff as compared to grass strips. Tree buffers established in combination with upstream grass strips perform better than buffers with trees alone.

Table 9.2 Sediment and nutrient trapping ability of riparian buffers

Species	Buffer width (m)	Trapping efficiency (%)				
		Sediment	Total N	Total P	PO_4-P	NO_3-N
Deciduous forest[1]	10	76	–	–	78	97
Hardwood trees and grasses[2]	75	–	27	56	56	59
Trees, shrubs, and grasses[3]	16	97	94	91	80	85
Shrubs and weeds[4]	18	30	32	30	–	60

[1]Schoonover et al. (2005), [2]Lowrance and Sheridan (2005), [3]Lee et al. (2003), and [4]Daniels and Gilliam (1996)

Fig. 9.4 Riparian buffers of A) trees and shrubs and B) trees and shrubs combined with native grass species (Courtesy USDA-NRCS)

9.5.1 Design of Riparian Buffers

Advanced ecological characterization techniques such as remote sensing, GIS, and mathematical models provide an opportunity to an effective planning, design, and establishment of riparian buffers across a wide spectrum of ecosystems. The complexity of agricultural landscapes requires the consideration of spatial analysis of site conditions and runoff patterns. Modeling buffers involves the analyses of runoff patterns and vegetation characteristics (e.g., width, growth pattern, density, canopy cover) to accommodate the site-specific soil conservation needs. A number of factors which must be considered in designing riparian buffers include the following:

1. ***Width***. The design width of strips is a function of plant species, land slope, and runoff rates. Current recommendation is that buffer width must be between 10 and 30 m. Narrow (<10 m) buffers are neither adequate to contain large sediment loads in runoff nor effective in improving the integrity wildlife habitat.
2. ***Vegetation***. Combination of forest species and grass species in wide strips (>10 m) is recommended to increase the ability of buffer strips for removing nutrients. Switchgrass, tall fescue, smooth bromegrass, and vetiver grass are some of the grass species used in riparian buffers. Native forest and grass species of contrasting ages, densities, and heights improve the performance of riparian buffers.

9.5.2 Ancillary Benefits

The new approach is that riparian buffers must not only reduce soil erosion and control transport of pollutants but also provide ancillary benefits including social and economic considerations (e.g. recreation, timber harvesting, C credits, wildlife habitat credits). Establishment of fast growing trees for fiber and biofuel production is a practical option in some agro-ecosystems to enhancing net income from buffer strip while still protecting the watercourses. Controlled harvesting of trees such as poplar is economically profitable (Henri and Johnson, 2005). Some forest riparian sites can benefit from moderate thinning and coppicing, depending on the forest species and growth stage. Threshold levels of harvesting of forest buffers without negatively affecting the functionality for erosion sediment control must be developed. Riparian buffers can also provide additional sources of income from the C credits and CRP. Traditionally, riparian buffers have been managed for water quality improvement rather than for C storage.

Expansion of these benefits can promote establishment of different scenarios of management of riparian buffers for enhancing net income on conservation management while improving quality of soil and water resources. The diversified use of buffers demands a careful planning and management of riparian systems.

9.6 Filters Strips

Vegetative filter strips are an area of grass (e.g., cool season grass) or other permanent vegetation planted between agricultural fields and streams for reducing sediment, nutrients, and other pollutants in water runoff to improve downstream water quality (Fig. 9.1). These buffers are commonly used in the USA and in some parts in Europe. The filter strips are a useful conservation practice to reduce water pollution with sediment, nutrients, heavy metals, and pesticides from agricultural fields. Under sheet flow, as much as 90% of sediment is reduced by 9-m wide filter strips (Fig. 9.5).

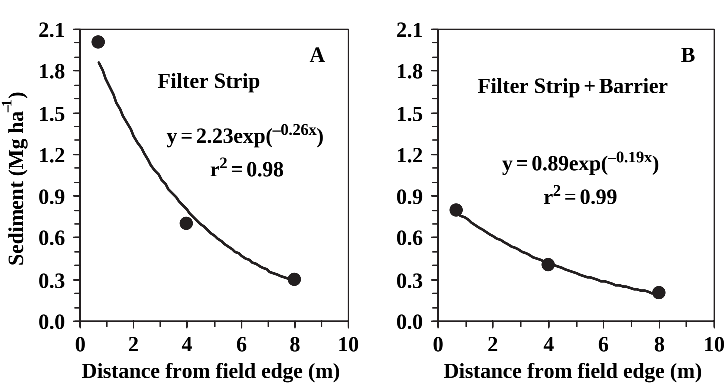


Fig. 9.5 Decrease in sediment mass with increase in width of tall fescue alone and in combination with switchgrass barriers (After Blanco-Canqui et al., 2004)

Sediment concentration decreases with increase in filter strip area. Most of the sediment and nutrients are trapped within the first few meters (2–3 m) of filter strips from the field boundary (Fig. 9.5). Reductions in nutrients by filter strips are smaller compared to reductions in sediments. Filter strips are effective for retaining runoff and sediments by increasing water infiltration into soil.

Example 1. Estimate the sediment trapping efficiency (STE) at 1 m below field edge for the filter strip systems in Fig. 9.5 assuming that the incoming sediment mass is 8.5 Mg ha^{-1} for both buffer systems. In addition, determine the amount of sediment trapped at the 6 m grass strip.

Solution:

1. **Sediment Trapping Efficiency**
   Tall fescue alone:

$$STE = \left(1 - \frac{Exiting}{Entering}\right) \times 100 = \left(1 - \frac{1.72}{8.5}\right) \times 100 = 79.8\%$$

Tall fescue plus 0.7 m Grass Barrier:

$$STE = \left(1 - \frac{0.72}{8.5}\right) \times 100 = 91.5\%$$

2. **Amount of Sediment Trapped**
   Tall fescue alone:

$$y = 2.23\, exp(-0.26x)$$
$$Trapping = 2.23\, exp\,(-0.26 \times 6) = 0.47\, Mgha^{-1}$$

Tall fescue plus 0.7 m Grass Barrier:

$$y \ = 0.89exp(-0.19x)$$
$$Trapping = 0.89exp\,(-0.19 \times 6) = 0.28\, Mgha^{-1}$$

9.6.1 Effectiveness of Filter Strips in Concentrated Flow Areas

The effectiveness of filter strips in trapping sediment and decreasing runoff rate and amount decreases under concentrated runoff (Fig. 9.6). The filter strips can fail under concentrated flow unless large filter strip areas are designed for dispersing the incoming flow and improving filter strip performance (Daniels and Gilliam, 1996). Filter strips are most effective in removing sediment and chemicals from uniform, shallow, and laminar overland flow. Concentrated runoff can overtop filter strips and reduce their sediment filtering capacity. The filter strips are particularly ineffective in soils with slopes >4%. Dispersion of concentrated runoff with drainageways is recommended to reduce the energy and velocity of runoff. The filter strips are

Fig. 9.6 Filter strip of tall fescue overtopped by concentrated flow (Courtesy of C.J. Gantzer, Univ. of Missouri, Columbia, MO)

effective to slow runoff, expand the runoff flow area, trap sediments and nutrients if the:

1. incoming runoff flow is uniform and laminar,
2. runoff rate is relatively small,
3. filter strips are wide enough (>10 m), and
4. filter strips are not inundated by runoff.

9.6.2 Grass Species for Filter Strips

In the USA, filter strips are often planted to cool season grasses including tall fescue, Kentucky bluegrass, orchard grass, smooth bromegrass, and others. Most of the cool season grasses were introduced into the USA from Europe, Asia, and Africa in the 1800's (USDA-NRCS, 1997a). The cool season grasses develop extensive and deep-root system allowing drought resistance and vigorous growth in early spring and late fall (Fig. 9.7). The most common species used in filter strips for erosion control is tall fescue. In temperate and subtropical climates, Bermuda grass, a warm season perennial species, is also used.

Tall fescue is a perennial cool season grass and reaches about 1 m in height. It is a bunchgrass and tends to form tight and dense sod. It produces short rhizomes and develops in sod-type growth. About 15 Mha of tall fescue are grown in the USA. Tall fescue is best adapted to the parts of the USA with hot and humid summers (Midwest, Mid-Atlantic, and Southeast). It is well adapted to a wide range of soils but grows best on clay soils, damp pastures, and wet environments. Tall fescue tolerates drought, surviving dry periods in a dormant state. It is more resistant to

Fig. 9.7 Cool season grasses used as filter strips (Courtesy USDA-NRCS)

low temperatures and can remain green later into the winter than other cool season grasses. It is a high yielding grass used widely for forage for late fall and winter grazing. Some varieties of tall fescue, however, may cause health problems in animals such as endophyte infestation, which decreases forage intake, growth, and production of milk. Because of both benefits and problems, the reactions of farmers to tall fescue are mixed and many are reluctant to its continued use.

Bermuda grass is a warm season, sod-forming, and perennial grass with deep and fibrous roots. It is best suited to erosion control in subtropical and tropical climates with 600–2,500 mm of annual precipitation and grows in many parts over the world. In the USA, it is mainly grown in southern and midwestern states. It withstands occasional drought periods but requires irrigation in arid climates. Bermuda grass grows well on a wide range of soils from clayey to sandy and can tolerate acidity and alkalinity and moderate waterlogging although it grows best in well-drained soils.

Kentucky bluegrass has a sod-forming ability and is well suited for soil erosion control especially when combined with other grass species. It can grow well in loamy or clayey soils with a pH between 5 and 7. Bluegrass reaches about 0.60 m in height, has fibrous root system, and resists overgrazing. It is mostly grown in north central and northeastern U.S. with temperatures below 24°C and is an ideal species for permanent pastures and for erosion control.

Orchard grass grows in clumps and forms sod. It is leafy grass with a fibrous and extensive system and adapted to a wide range of ecosystems in the USA particularly in the Appalachian Mountains, Midwest, and Great Plains. Orchard grass starts growth in early spring and flowers in late spring. Compared to Kentucky bluegrass, the root system is more extensive and deeper and so it is more resistant to drought. The optimum daytime temperature for growth of orchard grass is about 21°C, which is slightly lower than that for smooth bromegrass or tall fescue.

Smooth bromegrass is a long leafed and perennial species of about 1 m in height. It is one of the most useful cool season grasses for hay, pasture, and silage, and ground cover. It has been used for erosion control around ditches, waterways and gullies. It grows better on well-drained silt or clay loam soils. Because of its deep and extensive root system, smooth bromegrass is relatively resistant to drought.

9.7 Grass Barriers

Grass barriers are narrow strips of dense, tall, and stiff-stemmed perennial grass established perpendicular to the field slope to control soil erosion while decreasing slope width (USDA-NRCS, 1997b). Grass barriers differ from other grass strips because they possess stiff and robust stems adapted to local soil types and climates, and are commonly planted to native warm grass species.

Unlike other buffer strips, grass barriers:

- Decrease the field slope width by forming benches or natural terraces upslope of the grass barriers with time.
- Pond runoff above them reducing the kinetic energy of runoff and increasing the infiltration opportunity time of runoff water.
- Reduce the formation of ephemeral gullies by intercepting and dispersing the concentrated-type flow.

9.7.1 Natural Terrace Formation by Grass Barriers

Barriers established along the contour across swale-eroding areas can allow sediment deposition, forming a delta upslope from the barriers and developing natural mini-terraces with time (Fig. 9.8). A progressive leveling and filling of surface

Fig. 9.8 Grass barriers trap sediment above them (Courtesy Larry A. Kramer USDA-ARS, Deep Loess Research Station, Council Bluffs, Iowa)

depressions above barriers occurs with sediment. Barriers deposit sediment upstream from them, reduce the land steepness, and broaden the area for the runoff to slow and spread. Grass barriers are compatible with tillage systems by forming mini-terraces, if properly established and spaced. Average soil slope between the parallel barriers can be reduced by 10–30% due to gradual sediment buildup upslope of the barriers.

9.7.2 Runoff Ponding Above Grass Barriers

Runoff ponding above barriers is the major mechanism for the high effectiveness of grass barriers. Sediments progressively accumulate above barriers forming a delta. Vegetative debris lodges against the dense and robust grasses, and increases the hydraulic resistance of barriers, promoting deeper runoff ponding and higher sediment deposition. The depth of runoff ponding above barriers can be >0.4 m, and thus sediments are mainly trapped as a result of runoff retardation (Dabney et al., 1999). Relatively long residence time of runoff upslope from the grass barriers is essential for the barriers to reduce the sediment and runoff leaving the barriers. Runoff ponding retains about 90% of particles coarser than 0.125 mm and about 20% of particles <0.032 mm (Meyer et al., 1999). Theoretically, the more time the backwater remains ponded above the barriers, the more sediment with decreasing particles in size is deposited. The Stoke's law states that the settling velocity is proportional to the square of the particle diameter.

9.7.3 Use of Grass Barriers for Diverse Agroecosystems

Grass barriers are extensively used to control erosion around the world particularly in tropical regions. Barriers are more economical than traditional conservation practices (National Research Council, 1993). A common grass used as barriers in tropical soils is the vetiver grass. This grass may not survive in regions where temperature falls below $-15°$C. Thus, it is not commonly used in cold latitudes. In some countries, native plant species, which are perennial, stiff-stemmed, tall growing, dense, and tolerant to runoff inundation and sediment load are locally selected and used as barriers for the erosion control. Proper experimental selection and management procedures are pursued to identify grasses for reducing soil erosion and water runoff.

9.7.4 Use of Grass Barriers in the USA

The use of grass barriers is not new to the USA, but it has not been widely implemented as a conservation practice. Prior to settlement, extensive areas of the USA were under native warm season grasses including switchgrass, big bluestem, and Indian grass. These native grasses were, however, mismanaged by settlers and eventually replaced by introduced cool season grasses in many landscapes. Clearing lands for cultivation and overgrazing contributed to the mismanagement of warm season grasses. In recent years, there has been a renewed interest in re-establishing the warm season grasses because of their benefits to erosion control, soil C sequestration, wildlife habitat, and livestock forage.

The USDA federal and state conservation programs were initiated to investigate the grass barrier effectiveness for soil erosion control since the early 1990's (Kemper et al., 1992). The warm season grasses control sediment and runoff loss better than cool season grasses because they are taller and possess more rigid and stiffer stems. Since then, barriers have been promoted as a proven conservation practice for sediment and water pollution control. Farmers are, however, reluctant to adopt grass barriers for erosion control probably because of the long history of using cool season grasses in many parts of the country. The cool season grasses are relatively less expensive, more readily available, and more easily managed than warm season grasses. Vast areas of agricultural and nonagricultural lands have been converted to cool season grasses. There is little information about grass barrier effectiveness in controlling soil erosion. Barriers are mostly grown in isolated areas primarily for research purposes.

The warm season grass barriers have numerous advantages for improving wildlife habitat. The abundant biomass provide food for native ground animals. Because warm season grasses have more bare ground under the foliage and between the plants than cool season grasses, birds and other farm animals can use the native grasses for food, cover and nesting. Pheasants, quail, rabbits, songbirds, ducks, geese, wild turkeys, and muskrats are abundant in areas under native warm season grasses. The seeds are an important food resource for many birds and other animals. Short cool season grasses reduce the wildlife habitat. The warm season grasses remain tall because they are only grazed during summertime, and thus more ground cover is present under warm season grasses during and after the grazing season. These grasses also benefit the landowners by providing summer livestock forage during the time when cool season grasses are dormant.

9.7.5 Grass Species for Barriers: Vetiver grass

There are numerous grass species which are suitable as barriers (The Plants Database, 2007). Vetiver grass is a perennial, dense, tall, and stiff stemmed species native to South Asia (Grimshaw and Helfer, 1995). It forms thick barriers with a height of about 1.5 m and the root system $>$1m depth. Because of its non-invasive nature and high tolerance to drought, waterlogging, and overgrazing, vetiver grass is an ideal species for controlling water runoff and soil erosion across a wide range of ecosystems. It grows from elevations near sea level to about 2,500 m, under temperatures between -15 and 45°C, and precipitation between 200 and 6,000 mm (Greenfield, 2002). The tolerance to adverse conditions makes vetiver grass a suitable species to grow in marginal and reclaimed soils. While vetiver grass is used as fodder for livestock, it still stabilizes the soil because the crown of the plant grows below the soil surface. Vetiver barriers are planted in 0.50–1 m wide strips to minimize the land area under the barrier. Vetiver grass is the only species that is effective for controlling soil erosion on steep terrains (30 and 60% slope). Vetiver strips are also used as windbreaks.

A major challenge for the diffusion of vetiver technology worldwide is the poor planting and management of the grass. Farmers may be hesitant to adopt the vetiver technology for erosion control unless they obtain additional benefits from vetiver establishment. Communications about the side-benefits of vetiver grass buffers are needed to promote its adoption. Provisions for providing farmer training and establishment of demonstration sites are keys to the diffusion of the technology. There are only a few vetiver nurseries in erosion-affected areas to satisfy the large demands. Richard Grimshaw established the Vetiver Network to expand the use of vetiver grass for soil and water conservation around the world (http://www.vetiver.org). Consequently, vetiver grass is used around 90 countries in Asia, Africa, Australia, South America, Central America, and North America.

9.7.5.1 Switchgrass

Switchgrass is a native warm season grass 1–2 m tall, upright with bunch-type growth, perennial, and stiff-stemmed prairie grass mostly found in the Midwest and Great Plains native prairies in the USA (Fig. 9.9). It develops leafy growth and numerous upright stems. As it grows, more shoots emerge from the lower stems around the leaf nodes filling in gaps. It has deep and extensive roots which can penetrate to about 1.5 m depth. Switchgrass develops fine-textured flowers that bloom in late July or early August. Unlike the cool season grasses, switchgrass grows best in sunny summer days. It is an excellent species for slowly permeable soils (e.g., claypan soils) because it can tolerate poorly drained conditions. This grass is suitable for relatively acid soils with a pH ranging between 4 and 7.5 and a wide range of annual precipitation (400–2000 mm), and temperature (5°C–45°C). Switchgrass can also grow in dry-mesic environments with varying soil conditions. It provides

Big Bluestem **Switchgrass** **Indian grass**

Fig. 9.9 Some warm season grasses that are used as conservation buffers (Courtesy USDA-NRCS) good quality forage for livestock (Hintz et al., 1998). Switchgrass is an ideal species for soil erosion control, wildlife habitat improvement (nesting area, cover and food for birds), and permanent pasture. Because its production is high during summer, it supplies abundant forage for grazing when the growth of cool season grasses is limited. Slow establishment and management inconsistency have, however, limited its use.

9.7.5.2 Eastern Gamagrass

Use of eastern gamagrass for soil erosion control is not as common as switchgrass. It is often found in flood plains, along stream banks in the eastern U.S., and in some areas in the Midwest. Like switchgrass, it also has stiff stems and upright growth of 2–3 m tall. It has numerous short rhizomes with most of the leaves developing from the base of the plant. This highly productive grass is best adapted to deep soils and wet environments. Although its growth is slow during establishment, it starts growing early in spring and produces high quality forage when mature and is highly palatable, and very nutritious for livestock. Because of deep-rooting system, eastern gamagrass is tolerant to drought conditions besides withstanding poorly drained soils. Its bunch-type growth benefits wildlife habitat while reducing soil erosion. Eastern gamagrass is a recommended species by USDA-NRCS for restoring degraded lands.

9.7.5.3 Indian Grass and Big Bluestem

Indian grass is a perennial and 1- to 2-m tall bunch grass having rigid stems. It is a warm season grass found in most states in the USA and specifically in remnant native grass species sites. The Indian grass has short rhizomes and thus tends to spread slower than switchgrass. It is a good source of high quality forage during summer. Although it adapts well to different soils and environments, it requires moderately well drained soils. The Indian grass is a hardy grass and is easily established for reducing soil erosion and improving wildlife habitat (Hintz et al., 1998).

Big bluestem is a robust and perennial native bunch-type grass 1–2 m tall. It is thought to be more palatable than other warm season grasses when mature. Big bluestem tolerates drought conditions and soils with low water holding capacity. Its extended and deep roots enable it to tolerate drought. Big bluestem grows under a wide range of soils and environments. While establishment of big bluestem can be slow, it has appropriate characteristics for reducing soil erosion and enhancing wildlife habitat.

9.7.6 Grass Barriers and Pollutant Transport

Narrow (<1 m) barriers can trap between 50 and 90% of sediment in water runoff (Table 9.3). Grass barriers can also reduce losses of nutrients, pesticides, and other pollutants in surface runoff. The water pollutants trapped above and within the

Table 9.3 Pollutant trapping effectiveness of grass barriers and filter strips

Pollutant	Trapping efficiency (%)	
	Barriers	Filter strips
[1]Sediment	78	75
[1]Total N	51	41
[1]Total P	55	49
[1]PO_4-P	46	39
[2]Sediment	91	78
[2]Total N	67	55
[2]Total P	53	36
[2]PO_4-P	54	37
[2]NH_4-N	50	19

[1]Lee et al. (1998) and [2]Blanco-Canqui et al. (2004).

barriers can be biodegraded and transformed into simpler compounds, reducing their potential for pollution. Barriers can intercept debris and sediment in suspension to form massive filtering barriers. Barrier performance in concentrated flow depends upon the patterns of concentrated flow, field slope, type of grass, and size of the contributing area.

9.7.7 Design of Grass Barriers

Grass barriers are established in 0.75- to 1.2-m-wide strips (Kemper et al., 1992) commonly using tall warm season grasses. In comparison, filter strips are wider (5 to 15 m) and often planted to short-growing and cool season grasses (Dillaha et al.,

Fig. 9.10 Switchgrass barriers parallel to row crops (Courtesy Larry A. Kramer, USDA-ARS, Deep Loess Research Station, Council Bluffs, Iowa)

1989). Barriers can be used in combination with no-till and reduced tillage cropping systems. Grass barriers are often established on the contour in parallel rows at short intervals (<15 m) in the field (Fig. 9.10). The design of barriers contrasts with filter strips which are primarily grown along the bottom perimeter of croplands. Planted barriers must be narrow and dense to have enough strength to resist heavy loads of runoff and sediment. Barriers can be established form seeds or transplants. The USDA-NRCS has standard guidelines for establishing and managing grass barriers (USDA-NRCS, 2000).

9.7.8 Grass Barriers and Concentrated Flow

Grass barriers can reduce erosion caused by the concentrated flow and curtail the formation of ephemeral gullies by spreading the runoff upstream from the barriers. High stiffness and rigidity are important characteristics of barriers that enable them to better withstand the concentrated loads and ponding of backwater thus remaining upright longer than filter strips. Grass barriers may also help reduce the occurrence of concentrated flow past riparian buffers, and thus improve water quality. Grass barriers in combination with other buffer strips can thus be a cost-effective bioremediation option for removing pollutants. Dense and deep-rooted barriers enhance water infiltration into and through slowly permeable horizons and decrease the water runoff available to transport pollutants.

9.7.9 Combination of Grass Barriers with Other Buffer Strips

Grass barriers may perform better than other buffer strips for reducing the transport of sediment and nutrients in water runoff because they have stiff-stems and form dense strips. When combined with other practices, grass barriers can increase the effectiveness of:

- Vegetative filter strips by ponding and slowing runoff.
- Riparian buffers by dispersing the concentrated flow above the buffers.
- Grassed waterways by controlling the concentrated flow in critical points.
- Mechanically constructed terraces by improving the sediment deposition above the channels and stabilizing the terraces.

For example, filter strips can be as effective as grass barriers for low runoff rates, but they are not likely to be effective in concentrated flow areas. Because the filter strips possess less robust stems, they are easily submerged by concentrated type flow compared to grass barriers. Filter strips often trap less sediment and nutrients in runoff than barriers (Table 9.3). Thus, grass barriers can be a companion practice to improve the performance and effectiveness of filter strips when established across concentrated flow channels or gullies.

9.8 Grass Waterways

Grassed waterways are natural or constructed channels of dense and deep-rooted grass species established along the bottom perimeters of upland agricultural fields to drain and retard surface runoff while preventing formation of gullies and runoff erosion along the waterways (Fig. 9.11). Waterways are more widely used in the U.S. agricultural lands. Differences in land availability and ownership, land topography, climate, vegetation, and tillage and cropping systems influence the popularity of grass waterways. Dense and deep-rooted vegetation in waterways improves water infiltration, absorbs nutrients and pesticides, and filters and traps excess sediments while draining runoff flowing off the fields. The runoff and sediment transport control by the side-slopes of grass waterways is comparable to that of grass filter strips or grass barriers. Grassed channels reduce about 50% of herbicides compared to nongrassed waterways (Briggs et al., 1999). Well-established grass species not only stabilize the channels themselves but adjacent field edges, reducing scouring and gullying. Grassed waterways link fragmented agricultural or urban lands into a landscape mosaic improving the aesthetics and controlling soil erosion. Because of low traffic and dense vegetation, grass waterways also improve wildlife habitat.

Fig. 9.11 Grass waterways below corn fields (Courtesy USDA-NRCS)

9.8.1 Design

Standards for buffer establishment or design are often set on a rule-of-thumb basis, which may neither optimize the waterway dimensions nor incorporate local variations in topography and soil conditions. Grass waterways must be designed based on site-specific characteristics including the following:

1. ***Expected runoff volume and peak runoff storms***. Grass waterways must be designed properly so that they can withstand high peak runoff discharges and

reduce the transport capacity of runoff. Designs are often based on a 10-yr return period runoff storms. Grassed waterways must be designed so as to reduce the estimated velocity of the concentrated runoff.

2. ***Shape and size of waterways***. The size, width, shape, and length of the channels vary with local conditions. The width ranges between 10 and 50 m. Their cross-sectional area is often trapezoidal, parabolic or triangular, depending on the field topography. Parabolic channels with side slopes (4:1) are preferred over trapezoidal and triangular waterways to reduce earth movement and facilitate traffic and operations of farm equipment. Flat-bottomed channels spread concentrated runoff, increase runoff residence time, and enhance water infiltration more than V-shaped waterways (Fiener and Auerswald, 2005).
3. ***Selection of plant species***. Climate, topography, runoff characteristics, adaptability of species, and soil type determine the selection of species for waterways. Cool-season and sod-like forming species are mostly used for constructing waterways in the U.S. including Bermuda grass, tall fescue, Italian ryegrass, Kentucky bluegrass, and smooth bromegrass. Tall fescue is particularly used in temperate climates. Mixtures of native and warm season species such as switchgrass and big bluestem are also recommended. Plant species must have enough hydraulic roughness to resist peak runoff rates. The permissible runoff velocities along waterways range between 1 and 2 m s^{-1} for slopes $<$10% depending on the soil erodibility, vegetation type, and management (Haan et al., 1994). Mowed or cut grasses retard runoff much less than unmowed/uncut species. Mowed grasses may withstand velocities $<$1 m s^{-1} while good stand and unmowed grasses about 2 m s^{-1} for moderate field slopes ($\sim$4%).

Waterways are designed using the continuity equation and open channel flow theory.

$$Q = A \times V \tag{9.1}$$

where Q is rate of runoff ($m^3\ s^{-1}$), A is area of the waterway (m^2), and V is velocity of runoff (m s^{-1}) which is computed using the Manning's equation

$$V = \frac{R^{\frac{2}{3}} S^{\frac{1}{2}}}{\mathrm{n}} \tag{9.2}$$

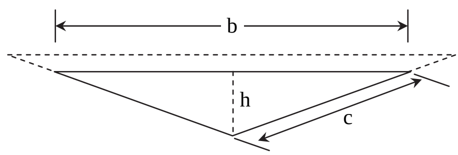
where R is waterway hydraulic radius (m), S is waterway slope (m m^{-1}), and n is vegetation roughness (Table 9.4). Based on the runoff velocity, optimum widths and depth of the waterways are proportioned by trial and error. A freeboard, distance from water surface to the top of channel, is included to account for changes in peak runoff rates.

Example 2. Calculate the velocity of runoff in a triangular waterway of 5 m base, 1 m height, and 4% slope under tall fescue. The grass is kept mowed to 0.12 m height.

Table 9.4 Roughness coefficients for selected grasses (After Haan et al., 1994)

Vegetation	Manning's roughness coefficient (*n*)
Alfalfa, sericea lespedeza, common lespedeza, and sudangrass	0.037
Native warm season grass	0.050
Tall fescue, and kentucky bluegrass, buffalo grass, blue grama, ryegrass (perennial), and bahiagrass	0.056
Bermuda grass and centipedegrass	0.074

First, calculate the hydraulic radius, which is defined as the area (A) of the flow section divided by the wetted perimeter (WP). The wetted diameter of a triangle is



$$\text{Area of a triangle, } A = b\left(\frac{h}{2}\right) \tag{9.3}$$

Using the Pythagorean theorem, the WP is computed as

$$c^2 = \left(\frac{b}{2}\right)^2 + h^2 \tag{9.4}$$

$$WP = 2\sqrt{\left(\frac{b}{2}\right)^2 + h^2} \tag{9.5}$$

$$Rh = \frac{bh}{4\sqrt{\left(\frac{b}{2}\right)^2 + h^2}} = 0.464\,\text{m}$$

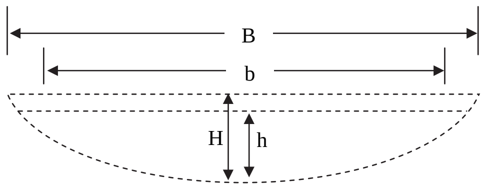
$$v = \frac{R^{\frac{2}{3}} S^{\frac{1}{2}}}{n} = \frac{(0.464)^{\frac{2}{3}}\,(0.04)^{\frac{1}{2}}}{0.25} = 0.479\,\text{ms}^{-1}$$

Example 3. Determine the dimensions of a parabolic grass waterway under a mixture of native and warm season grass to safely convey 4 $\text{m}^3\ \text{s}^{-1}$ of surface runoff on a silt loam soil with a 5% slope. Assume velocity of runoff at 1.8 m s^{-1}.

Using the continuity equation, the area of the waterway is

$$A = \frac{Q}{V} = \frac{4\,m^3 s^{-1}}{1.8\,ms^{-1}} = 2.22\,m^2$$

Cross-sectional area of a paraboloid:



$$A = \frac{2bh}{3} \tag{9.6}$$

$$WP = b + \frac{8h^2}{3b} \tag{9.7}$$

$$R = \frac{2h}{3} \tag{9.8}$$

$$\text{Top width} = B = b\left(\frac{H}{h}\right)^{\frac{1}{2}} \tag{9.9}$$

$$b = \frac{A}{0.67h} \tag{9.10}$$

Calculate the hydraulic radius of the required waterway

$$R = \left(\frac{Vn}{S^{\frac{1}{2}}}\right)^{\frac{3}{2}} = \left(\frac{1.8 \times 0.025}{(0.05)^{\frac{1}{2}}}\right)^{\frac{3}{2}} = 0.09\,\text{m}$$

Then, the height of the waterway is

$$h = \frac{3R}{2} = \frac{3 \times 0.09}{2} = 0.135\,\text{m}$$

The top width (b) of the waterway is

$$b = \frac{A}{0.67h} = \frac{2.22\,m^2}{0.67 \times 0.135\,m} = 24.54\,\text{m}$$

A freeboard between 0.10 and 0.15 should be added to b to compute final design specifications

$$h_{final} = h + h \times 0.15 = 0.155\,m$$
$$B = b + b \times 0.15 = 28.22\,m$$

Verify whether the new dimensions can carry 4 $\text{m}^3\ \text{s}^{-1}$ of incoming runoff

$$Q = A \times V = \frac{2bh}{2} \times V = \frac{2 \times 24.54 \times 0.135}{3} \times 1.8\,ms^{-1} = 3.98\,m^3s^{-1}$$

The resulting design dimensions are appropriate for the specified conditions. Further detailed discussions about the engineering designs of soil erosion control structures are presented in textbooks of agricultural engineering (e.g., Schwab et al., 1993).

9.8.2 Management of Waterways

Grass waterways require adequate maintenance to reduce runoff and soil erosion. Sedimentation diminishes the ability of waterways to function as runoff drainageways and control gully formation. Grass waterways must be mowed to maintain the appearance and reduce proliferation of weeds. Excessive mowing can, however, reduce the hydraulic roughness and the filtering ability, causing swamping of grasses under concentrated runoff. Uncontrolled traffic and mowing compact the waterways and reduce water infiltration. Grassed waterways perform the best when combined with grass filter strips or terraces above. Terraces break the field slopes and reduce the scouring capacity of the incoming runoff while filter strips trap most of the sediment leaving the field and reduce sedimentation of downstream channels. The rapid expansion in agriculture and urbanization has raised concerns over the proper dimensions and maintenance of grass waterways. Grass waterways must be as continuous and wide as possible to maximize their effectiveness. Establishing grass waterways at the watershed-scale for linking broken fields and habitats is recommended over single or isolated waterways.

9.9 Field Borders

Field borders are narrow bands of perennial vegetation established around or at the edge of farm lands (Fig. 9.12). They reduce erosion by water and wind, improve the air quality, increase the C biomass input, reduce incidence of invasion of insects to farms, and improve farm aesthetics. Field borders are often established around road ditches and grassed waterways and differ from other buffer strips in that they are established both uphill and downhill sides of the field. For example, filter strips and grass barriers are commonly placed on the downhill end of the field to remove sediment, nutrients, and pesticides from runoff. Field borders when placed at the bottom of fields also serve as filter strips. Borders reduce losses of sediment and nutrients between 30 and 75% (Vache et al., 2002).

Unlike most buffer strips, field borders are sometimes used as a strip to turn farm equipment, travel around fields, and access to crop fields. Adequately managed field borders conserve soil and water and stabilize the cropland perimeters in spite of the constant wheel traffic. Width of farm machinery and amount of traffic are factors to be considered in designing field borders. Tall and robust plant species with stiff

Fig. 9.12 Field borders used for vehicular traffic (Courtesy H. Blanco)

stems including warm season grasses, native forbs, and shrubs are recommended for field borders. Deep-rooted and sediment-tolerant species must be used when borders are placed near waterways or stream channels to stabilize banks. If possible, field borders must be established around the whole perimeter of croplands to provide effective erosion control. Establishing borders long enough and perpendicular to the prevalent wind direction is important to reducing wind erosion. Species that produce large amounts of biomass are recommended to enrich the soil organic matter.

9.10 Modeling of Sediment Transport through Buffer Strips

Models for predicting the effectiveness of buffer strips for erosion control are based upon the principles of the sediment continuity equation, and their approach is that runoff flow loses its transport capacity when it enters the buffer strip area, causing deposition of sediment. Deposition is directly proportional to the transport capacity of runoff and sediment load and it occurs when the runoff transport capacity is less than the sediment load. Most of the sediment is deposited within the upper portions of the buffer strips.

Models are developed based on the following assumptions:

- Homogeneous sediment transport
- Tractive force is less than critical value for the channel
- Quiescent settling conditions in turbulent free flow
- Uniform flow velocity distribution,
- No lateral sediment inflow
- Settling velocity is the same for particles of the same size
- Sediment spread evenly throughout the depth of flow
- Deposition rate decreasing linearly downstream

- Unaltered bed slope and all sediment reaching the bed is trapped
- Steady-state flow rate
- Vegetation remains rigid and not submerged by concentrated flow

Some of these assumptions do not reflect the sediment transport behavior under field conditions. For example, buffer strips may fail under submerged flow decreasing significantly the roughness and hydraulic resistance to runoff with time.

The sediment prediction through grass buffers requires the identification of sediment deposition zones. Figure 9.13 illustrates a combined grass buffer system where switchgrass is grown immediately above tall fescue filter strips. The diagram shows three zones (A, B, and C) where the process of sediment deposition pattern varies as dictated by grass type. Sediment deposition and transport is predicted separately through each zone. The area upslope of the grass strips is the zone A in which runoff ponding takes place. Zone B confines the grass barriers, while Zone C is the area within the filter strip where transport of bedload sediment is zero and thus sediment is mainly transported in suspension.

Barfield et al. (1979) identified four sediment deposition zones (A, B, C, and D) where Zone A is the area immediately upslope of the grass strips, Zone B is the zone of major sediment deposition found between Zones A and C, Zone C is a transition and narrow portion of the grass strips having little bedload sediment transport but higher than that in Zone D, and Zone D is the longest section of the strip. Runoff in Zone D mainly carries fine and colloidal particles in suspension. Bedload sediment transport is reduced to minimum amounts as the distance of the grass strip increases.

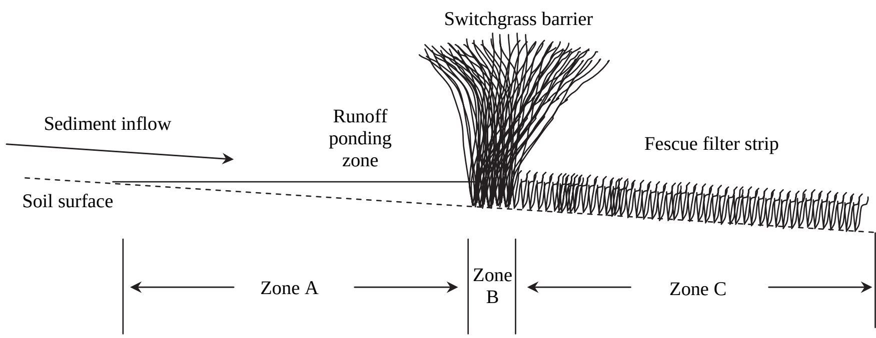


Fig. 9.13 Sediment deposition zone above the switchgrass barriers and the sediment filtering zone within the filter strip (Not to scale)

9.10.1 *Process-Based Models*

Process-based models such as the Vegetative Filter Strip Model (VFSMOD), Riparian Ecosystem Management Model (REMM), SWAT, GLEAMS, and WEPP can be used to estimate the transport of pollutants through buffers from the field edge to the water streams (Abu-Zreig et al., 2001; Munoz-Carpena and Parsons, 2004). These models simulate the effects of different buffer management scenarios on sediment and nutrient transport based on sensitive input parameters including vegetation and soil type, upland management, field slope, and climate conditions. The VFSMOD is specifically designed to test the filter strip performance for reducing sediment transport by using three submodels: modified Green-Ampt, overland flow, and kinematic wave approximation. Most models indicate that field slope, soil conditions, and runoff characteristics are the most sensitive parameters affecting the effectiveness of buffers.

9.10.2 Simplified Equations

9.10.2.1 Equation 1

Tollner et al. (1976) derived a simplified equation to predict sediment trapped (ST) by grass buffers based on two independent variables: 1) particle fall number (N_f) and 2) turbulent Reynold's Number (R_e) for different particle size fractions as

$$ST(\%) = 44.1 N_f^{0.29} \tag{9.11}$$

$$ST(\%) = 4.1^{*}10^{2}\,(R_e)^{-0.28} \quad \text{diameter of particles} = 0.47\,\text{mm} \tag{9.12}$$

$$ST(\%) = 1.1^{*}10^{6}\,(R_e)^{-1.96} \quad \text{diameter of particles} = 0.067\,\text{mm} \tag{9.13}$$

$$ST(\%) = 6.0^{*}10^{5}\,(R_e)^{-2.07} \quad \text{diameter of particles} = 0.027\,\text{mm} \tag{9.14}$$

$$R_e = \frac{V_m R_s}{\upsilon} \tag{9.15}$$

$$N_f = \frac{V_s L}{V_m d_f} \tag{9.16}$$

$$V_m = \left(\frac{1.0}{xn}\right) R_s^{\frac{2}{3}} S_c^{\frac{1}{2}} \tag{9.17}$$

$$R_s = \frac{S_c d_f}{2d_f + S_c} \tag{9.18}$$

where V_m is Manning's flow velocity (m s^{-1}), R_s is hydraulic radius of the channel (m), υ is water kinematic viscosity (s m^{-2}), V_s is particle settling velocity (m s^{-1}), L is width of the grass strip (m), d_f is flow depth (m), xn is a calibrated value for Manning's roughness of the grass strips, and S_c is slope of the flow channel (m m^{-1}).

The combined equation (Tollner et al., 1976) is

$$\frac{S_i - S_o}{S_i} = exp\left[-A\frac{(R_e)^B}{\left(N_f\right)^C}\right] \tag{9.19}$$

where S_i is inflow of sediment, S_o is outflow of sediment, A, B, and C are constants having the following values: A $= -1.05^{*}10^{-3}$, B $= 0.82$, and C $= -0.91$ determined from experimental data.

9.10.2.2 Equation 2

Foster (1982) formulated two approaches based on the model by Tollner et al. (1976). The first approach neglects the sediment ponding upslope of the strips and assumes that sediment deposition begins at the upper edge of the grass strip. The transport capacity (T_c) and depth of deposition (Y_d) at given intervals within the grass strip are calculated as

$$q_{so} = T_c - (T_c - q_{sin})\, exp\,[-\alpha L_w] \tag{9.20}$$

$$Y_d = \left(\frac{1}{\rho_d}\right) \int D_r dt \tag{9.21}$$

where q_{so} is sediment outflow rate (m s^{-1}), q_{sin} is sediment inflow rate (m s^{-1}), α is first order reaction coefficient for deposition, L_w is width of the strip (m), ρ_d is effective density of the deposited sediment (Mg m^{-3}), Dr is deposition rate, and t is time (s).

The second approach includes the analysis of sediment ponding. Deposition of sediment occurs in the pond and extends upstream from the strip starting as a concave slope until it becomes almost linear. The sediment leaving the pond (q_{so}) and the volume of sediment deposited in the pond upslope the strip (V_{dp}) in a given time interval (Δt) are computed as

$$q_{so} = q_{sin} exp(-\alpha L p) \tag{9.22}$$

$$V_{dp} = \frac{\Delta t(q_{sin} - q_{so})}{\rho_d} \tag{9.23}$$

where L_p is the length of the pond.

9.10.2.3 Equation 3

Flanagan et al. (1989) simulated the sediment deposition (D_F) as

$$D_F = \left[\frac{\varphi}{(1+\varphi)}\right]\left(\frac{dT_c}{dx} - D_L\right)\left[1 - \left(\frac{x_u}{x}\right)^{1+\varphi}\right] + D_u\left(\frac{x_u}{x}\right)^{1+\varphi} \tag{9.24}$$

$$\varphi = \frac{\beta v_f}{\sigma} \tag{9.25}$$

where φ is measure of deposition, T_c is flow transport capacity at x, a position downslope, D_L is lateral sediment inflow, x_u is width of source area, D_u is deposition rate at x_u, β is turbulence factor, v_f is particle fall velocity, and σ is the rainfall rate. The sediment load (G) in the grass strip is calculated as

$$G = \int (D_F + D_L)dx \tag{9.26}$$

Substitution of Eq. (9.24) into Eq. (9.26) yields

$$G = \left[\frac{\varphi}{(1+\varphi)}\right]\left(\frac{dT_c}{dx}\right)\left[x + \left(\frac{x_u^{1+\varphi}}{\varphi}\right)x^{-\varphi}\right] - D_u\left(\frac{x_u^{1+\varphi}}{\varphi}\right)x^{-\varphi} + C \tag{9.27}$$

C is integration constant. This expression for non-uniform sediment delivery ratio, *SDR*, composed of a wide range of sediment particles reduces to

$$SDR = \sum_{i=1}^{5} f_i x_u^{\varphi i} \tag{9.28}$$

where f_i is fraction of particle type i entering the strip. Five particle types are: primary clay, primary silt, small aggregates, large aggregates, and primary sand. The simplified equation for estimating the sediment trapping efficiency, *STE* is

$$STE = (1 - SDR)100 \tag{9.29}$$

9.10.2.4 Equation 4

Haan et al. (1994) compiled various sediment transport equations derived by Tollner et al. (1976), Barfield et al. (1979), and Hayes et al. (1984) into a workable assembly and provided examples on how to analytically compute the: (1) runoff velocity and depth and (2) sediment transport efficiency in grass buffer strips. Runoff flow velocity and depth, average size distribution of sediment particles, bedload sediment transport capacity, settling velocities, particle fall number, Reynold's Number, and the Manning's roughness coefficient are the main factors to estimate the sediment trapping efficiency. A critical value in the estimation of flow velocity is the Manning's roughness coefficient. Haan et al. (1994) proposes the use of calibrated values of Manning's roughness to estimate the runoff velocity. The calibrated coefficient of Manning's roughness is 0.056 for tall fescue, blue grama, perennial ryegrass, Kentucky bluegrass, and bahiagrass, 0.074 for bermuda grass and centipedegrass, and 0.05 for other grass mixtures.

9.10.2.5 Runoff Velocity and Depth

Runoff flow velocity (V) through buffer strips is computed using a calibrated form of Manning's equation

$$V = \frac{R_s^{\frac{2}{3}} S_c^{\frac{1}{2}}}{xn} \tag{9.30}$$

where S_c is channel slope, xn is the calibrated value of Manning's coefficient of roughness (R_s) is the hydraulic radius of the media spacing of the buffer strip as

$$R_s = \frac{S_s d_f}{2d_f + S_s} \tag{9.31}$$

where S_s is grass strip spacing (m). The flow per unit width of grass strip (q_m) is estimated as

$$q_m = V_m d_f \tag{9.32}$$

Example 4. Estimate the runoff depth and velocity across a 10-m wide tall fescue filter strip established below a moldboard plowed cropland field on a Mexico silt loam with 4.5% slope if the runoff flow rate is $6.86 \times 10^{-3}\ \mathrm{m^3\,s^{-1}}$ per unit width of grass strip. Assume S_s equal to 0.015 m.

Runoff depth:

$$q_m = d_f \frac{R_s^{\frac{2}{3}} S_c^{\frac{1}{2}}}{xn} = 6.86 \times 10^{-3} = \frac{d_f^{\frac{5}{3}}}{0.056} \left(\frac{0.016}{2d_f + 0.016} \right)^{\frac{2}{3}} (0.045)^{\frac{1}{2}}$$

The solution for d_f by trial and error is: $d_f = 0.05\,\mathrm{m} = 5\,\mathrm{cm}$

Runoff velocity:

$$R_s = \frac{S_s d_f}{2d_f + S_s} = \frac{0.016 \times 0.05}{2 \times 0.05 + 0.016} = 0.0069\,\mathrm{m} = 0.69\,\mathrm{cm}$$

$$V_m = \frac{R_s^{\frac{2}{3}} S_c^{\frac{1}{2}}}{xn} = \frac{(0.0069)^{\frac{2}{3}} (0.045)^{\frac{1}{2}}}{0.056} = 0.137\,\mathrm{m}$$

Check the flow rate using the estimated d_f and V_m:

$$q_m = V_m d_f = 0.137 \times 0.05 = 6.85 \times 10^{-3} ms^{-1}$$

9.10.2.6 Sediment Trapping Efficiency

The fraction of soil particles entering the grass strips is computed as follows (Hayes et al., 1984):

$$a_v = f_{ri} + \frac{f_{ri}^{11}}{2} \tag{9.33}$$

where a_v is average size distribution, f_{ri}^{11} is coarse material fraction, and f_{ri} is fraction of particles <0.04 mm. The bedload transport capacity is equal to

$$q_{sd} = \left(\frac{K(R_s S_c)^{3.57}}{d_{pd}^{2.07}} \right) \tag{9.34}$$

$$K = 6.462^{*} 10^{7} \rho_s (\rho_s - 1)^{-3.07} \tag{9.35}$$

where q_{sd} is amount of sediment deposited (g s^{-1} cm^{-1}), d_{pd} is particle diameter (mm), and ρ_s is particle density (g cm^{-3}). Sediment deposition above the grass strips (f) is predicted as

$$f = \frac{q_{si} - q_{sd}}{q_{si}} \tag{9.36}$$

where q_{si} is incoming sediment load of coarse material (>0.037 mm) (f_{ri}^{c}).

$$q_{si} = q_{si} f_{ri}^{c} \tag{9.37}$$

Particle size distribution (D_{rd}) leaving the grass strips is computed as

$$D_{rd} = 1 - f^{*} f_{ri}^{11} \tag{9.38}$$

The settling velocities (V_s) and outflow rate of runoff (q_{wo}) are estimated as

$$V_s = 2.81 d_p^2 \tag{9.39}$$

$$q_{wo} = q_{wd} - iL \tag{9.40}$$

where d_p is average diameter of medium size particles, q_{wd} is inflow rate of runoff, i is infiltration rate, and L is width of grass barrier or filter strip. The fraction of sediment trapped by settling (T_s) is

$$T_s = exp\left[-1.05^{*} 10^{-3} R_e^{0.82} N_f^{-0.91} \right] \tag{9.41}$$

$$\text{or } T_s = \frac{q_{sin} - q_{so}}{q_{sin}} = exp\left(-1.05x10^{-3} R_e^{0.82} N_f^{-0.91} \right) \tag{9.42}$$

where q_{sin} is incoming sediment load, and q_{so} is sediment load leaving the buffer strips. The infiltration parameter (I) and fraction trapped in the litter (f_d) are

$$I = \frac{q_{wd} - q_{wo}}{q_{wd} + q_{wo}} \tag{9.43}$$

$$f_d = \frac{T_s + 2i(1 - T_s)}{1 + i(1 - T_s)} \tag{9.44}$$

where q_{wd} and q_{wo} are the flow rates entering and exiting the grass strip. Flow channelization within filter strips is corrected as

$$C' = 0.5exp\left[-3D_{ep}\right] + 0.5exp\left[15(0.2D_{ep} - D_{ep}^2)\right] \tag{9.45}$$

$$T_s = C'T_s \tag{9.46}$$

where C' is the correction function to account for the particles reaching the bed, T_{cs} is the corrected trapping efficiency, and D_{ep} is the average depth of sediment deposited. Finally, the equation for estimating the total sediment trapping efficiency (f_{to}) is computed as

$$f_{to} = \left[f + f_{d-sand}(1-f)\right]\left(1 - f_{ri}^1\right) + f_{d-silt}\left(f_{ri}^1 - f_{ri}^0\right) + f_{d-clay}f_{ri}^0 \tag{9.47}$$

where f is the fraction trapped as bedload sediment in zone A, and f_{d-sand}, f_{d-silt}, f_{d-clay} are the soil fractions trapped in zone B. The f_{ri}^0 is the fraction of inflow sediment smaller than 0.002 mm. A suggested correction factor (C_{cf}) for the channelized flow to the inflow rate (q_{wi}) and incoming sediment load (q_{si}) is

$$q'_{wi} = q_{wi}/C_{cf} \tag{9.48}$$

$$q'_{si} = q_{si}/C_{cf} \tag{9.49}$$

For example, if flow occurs only through 50% of the grass strip area, the C_{cf} factor is 0.5.

Example 5. Calculate the sediment trapping efficiency for the tall fescue filter strip in Example 4 using the approaches by Haan et al. (1994). Assume the settling velocities within the filter strip equal to 6.3×10^{-3} m s^{-1} for sand, 1.215×10^{-4} m s^{-1} for silt, and 3.372×10^{-6} m s^{-1} for clay particles. The infiltration rate of the soil is 1.1167×10^{-6} m s^{-1}. The fraction trapped as bedload sediment at the field edge is 0.38 with a silt fraction equal to 0.30 and clay fraction 0.15.

Estimate the Reynold's number and the fall number

$$R_e = \frac{V_m R_s}{\upsilon} = \frac{(0.137)(0.0069)}{9.77\times10^{-7}} = 967.55$$

$$N_f = \frac{V_s L}{V_m d_f} = \frac{V_s \times 10}{0.137\times0.05} = 1459.85V_s$$

Compute fall number for each particle size fraction

$$\text{Sand} = 1459.85\times6.3\times10^{-3} = 9.197$$

$$\text{Silt} = 1459.85\times1.215\times10^{-4} = 0.177$$

$$\text{Clay} = 1459.85\times3.372\times10^{-6} = 0.00492$$

The infiltration parameter for the system is

$$I = \frac{q_{wd} - q_{wo}}{q_{wd} + q_{wo}} = \frac{1.1167 \times 10^{-6}}{6.86 \times 10^{-3}} = 1.628 \times 10^{-4}$$

The fraction of sand, silt, and clay trapped by settling within the filter strip is $T_s = exp\left[-1.05^{*}10^{-3} R_e^{0.82} N_f^{-0.91}\right]$

$$T_s \text{ for sand} = exp\left[-1.05^{*}10^{-3}(967.55)^{0.82}(9.197)_{-0.91} = 0.962\right]$$
$$T_s \text{ for silt} = exp\left[-1.05^{*}10^{-3}(967.55)^{0.82}(0.177)_{-0.91} = 0.240\right]$$
$$T_s \text{ for clay} = exp\left[-1.05^{*}10^{-3}(967.55)^{0.82}(0.00492)^{-0.91} = 7.47 \times 10^{-17}\right]$$

The total amount of sediment trapped accounting for the water infiltration is

$$f_d = \frac{T_s + 2i(1 - T_s)}{1 + i(1 - T_s)} = \frac{\mathrm{T_s} + (2)(1.628 \times 10^{-4})(1 - \mathrm{T_s})}{1 + (1.628 \times 10^{-4})(1 - \mathrm{T_s})}$$

$$f_{d-sand} = \frac{0.962 + (2)(1.628 \times 10^{-4})(1 - 0.962)}{1 + (1.628 \times 10^{-4})(1 - 0.962)} = 0.962$$

$$f_{d-silt} = \frac{0.24 + (2)(1.628 \times 10^{-4})(1 - 0.24)}{1 + (1.628 \times 10^{-4})(1 - 0.24)} = 0.240$$

$$f_{d-clay} = \frac{7.47 \times 10^{-17} + (2)\left(1.628 \times 10^{-4}\right)\left(1 - 7.47 \times 10^{-17}\right)}{1 + \left(1.628 \times 10^{-4}\right)\left(1 - 7.47 \times 10^{-17}\right)} = 0.000326$$

Finally, the total sediment trapping efficiency of the 10-m filter strip is computed as

$$\mathrm{STE} = \left[f + f_{d-sand}(1 - f)\right]\left(1 - f_{ri}^{1}\right) + f_{d-silt}\left(f_{ri}^{1} - f_{ri}^{0}\right) + f_{d-clay}\, f_{ri}^{0}$$
$$\mathrm{STE} = [0.38 + 0.962\,(1 - 0.38)]\,(1 - 0.30) + 0.24\,(0.30 - 0.15)$$
$$+ 0.000326 \times 0.15 = 0.72$$

Thus, the 10-m wide tall fescue filter strip can reduce sediment loss by 72%. Additional details of computation of sediment trapping efficiency within filter strips are provided by Haan et al. (1994).

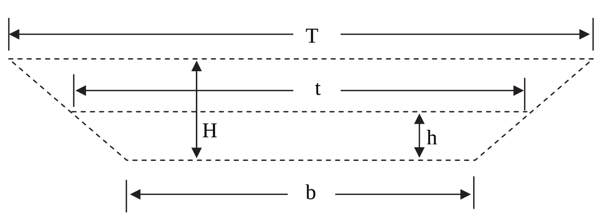
Summary

Buffer strips are permanent vegetation strips established perpendicular to the field slope to intercept, trap, and remove sediment and chemicals in runoff from agricultural lands. Buffer strips improve water infiltration and soil structural properties and increase C storage. Buffers stabilize streambanks and improve wildlife habitat recovery and protection. They remove sediment and nutrients by ponding runoff, promoting sediment deposition, increasing water infiltration, and increasing immobilization and transformation of chemicals. Factors that affect performance of buffers include: runoff velocity and rate, flow channelization, vegetation type, width of strips, soil particle size, soil structural characteristics, soil slope, upland management, and size of sediment source area. The main types of buffer strips are: riparian buffers, filter strips, grass barriers, grassed waterways, field borders, and windbreaks.

Removal of sediment and nutrients in buffer strips increases with increase in the width of buffer strips. Most of the sediment is deposited within the upper portions of the grass strip. Filter strips are effective for reducing sediment transport in shallow flow but may not be as effective under concentrated flow. Filter strips are often planted to cool season grasses, while grass barriers are narrow strips of stiff-stemmed perennial planted to vetiver grass, switchgrass, big bluestem, indian grass, and eastern gamagrass. These barriers pond runoff, filter sediment, and form miniterraces over time. A number of models have been developed to predict the effectiveness of buffers to filter sediment and nutrients based on the specific characteristics of vegetation, slope, and management.

Study Questions

1. Discuss the mechanisms by which buffer strips reduce runoff and soil erosion.
2. Describe the mechanisms of nutrient removal in runoff by buffer strips.
3. Compare filter strips with grass barriers in terms of design and effectiveness.
4. Compare the magnitude of increases in runoff velocities through a trapezoidal waterway seeded to unmowed tall fescue with a bottom width of 4 m and side slopes of 4:1, if the soil slopes along the landscape change changes from 1, 4, and 5%.
5. Compare results from Prob. 4 with those from a waterway under bare soil but with the same slopes.
6. Compute the dimensions of a trapezoidal waterway under mowed and unmowed native warm season grass to transport 8 $m^3\ s^{-1}$ of runoff on a soil with 3% of slope. The side slopes of the waterway are 4:1. Assume a permissible velocity of 1.2 $m\ s^{-1}$. Add the necessary freeboard.



7. Design the dimensions for a parabolic waterway under Bermuda grass to carry 6.5 $m^3\ s^{-1}$ on a soil with 2% slope.

8. Estimate the width of tall fescue filter strip needed to reduce sediment loss from 12 Mg ha^{-1} to 4 Mg ha^{-1} on a 5% soil slope.
9. Estimate the amount of sediment that is retained at 6 m filter strips below the field edge receiving 8.5 Mg ha^{-1} of sediment load. Use the empirical relationships in Fig. 9.5.
10. Estimate the runoff velocity and the sediment trapping efficiency for a 6-m wide bermuda grass on a soil with 5% of slope if the runoff flow rate is 5.50×10^{-3} m^3 s^{-1} per unit width of grass strip for Example 5. The infiltration rate of the soil is 3.1167×10^{-6} m s^{-1}.

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Chapter 10
Agroforestry

Agroforestry is a land management system that combines trees and/or shrubs with agricultural crops and livestock production on the same piece of land (Fig. 10.1). It is an emerging technology for effective soil and water conservation. In a broader sense, agroforestry comprises a wide range of practices that involve establishing and managing trees intentionally around or within croplands, pasture lands, and farm animal grounds with the purpose of controlling soil erosion, developing sustainable agricultural production systems, improving wildlife habitat and rural landscape, mitigating environmental pollution, and increasing farm economy through harvesting of tree-based specialty products.

Fig. 10.1 Agroforestry practices reduce water and wind erosion in nearly level soils in Shelby County, Missouri (Courtesy of Ranjith Udawatta, Center for Agroforestry at the Univ. of Missouri)

H. Blanco, R. Lal, *Principles of Soil Conservation and Management*,

10.1 Importance

Agroforestry systems are among the innovative options to manage and conserve soil and water, restore soil fertility, and halt desertification. These systems not only control soil erosion but also provide tree-based marketable products. They are principally essential to regions with high rainfall intensities, steep slopes, and sparse vegetation with high rates of runoff and soil erosion. Agroforestry fits within the environmental stewardship of soil, water, and air improvement towards a sustainable management of natural resources. Combining trees and/or shrubs with traditional crops is a biological and ecological approach to halt water and wind erosion. This system is an alternative to costly soil erosion control structures (e.g. terraces) for reducing runoff and soil erosion.

10.2 Classification

The numerous agroforestry systems can be grouped into five major practices (Young, 1997):

- alley cropping
- forest farming
- silvopasture
- riparian forest buffer
- windbreaks

This Chapter discusses only the first three systems (alley cropping, forest farming, and silvopasture). Discussions about windbreaks and riparian forest buffers are presented in Chapters 3 and 9.

10.3 History

Agroforestry has long been in use around the world particularly in tropical and subtropical climates. It has been used since ancient times in Asia, Africa, and precolonial America. Records show that the Inca civilization in South America practiced agroforestry for fuel and timber as early as AD 1100 (Chepstow-Lusty and Winfield, 2000). The long agroforestry tradition in the Andean regions was tailored with the arrival of colonial times in the 1500'and new tree species (e.g., Eucalyptus) were introduced to highland ecosystems. Today, there is a trend to restore old agroforestry approaches by using native tree and shrub species as a means for controlling soil erosion in prime and degraded lands. The International Center for Research in Agroforestry (ICRAF), now World Agroforestry Center, is engaged in the development and expansion of agroforestry projects in the tropics and sub-tropics in Africa and other developing regions.

The use of agroforestry in temperate zones is recent. In the Americas, following substantial clearance of native forest between 1600s and 1800s, new tree species were introduced to reforest cleared lands. Establishing permanent trees around agricultural lands in the USA was not, however, very common until the Great Depression and Dust Bowl era during 1930s, which increased interest and research in soil conservation practices. The USDA National Agroforestry Center (NAC), created in 1990 Farm Bill, stimulated research on the design, management, and transfer of technologies for alley cropping, forest farming, riparian buffers, silvopasture, and windbreaks (USDA-NAC, 2006).

10.4 Current Trends

Agroforestry practices, originally conceived to address problems of water and wind erosion, are now being increasingly used to reduce non-point source water pollution in developed countries. Coincidently, there is also an emerging emphasis on the potential of agroforestry systems for mitigating the projected changes in global climate through C sequestration and reduction of net emissions of greenhouse gases to the atmosphere. In the developing world, agroforestry is a useful practice for alleviating poverty and advancing food security. In many African countries, introduction of agroforestry practices to rural communities has substantially improved farm economy and reduced hunger by improving soil fertility and hence increasing crop production (Garrity, 2004). Agroforestry also generates employment by growing crops and marketing the trees and tree-derived products (e.g., fruit, latex, resins, nuts, ginseng, timber, berries, medicinal products) (Leakey et al., 2003).

Agroforestry is receiving greater attention and wider recognition as a solution to several and contrasting environmental, social, and economic problems around the world. Agroforestry technology is a holistic approach and an innovative option to harness benefit to mitigate both the poverty and the projected global climate change. In the past, adoption of agroforestry was considered as an optional means to conserve soil and water and improve land aesthetics or diversify farming. Presently, adoption of these systems is a necessity to help reduce the poverty and mitigate atmospheric and water pollution. Despite numerous benefits, adoption of agroforestry systems is still limited worldwide.

10.5 Functions of Agroforestry

Agroforestry provides innumerable environmental and ecological benefits and are thus multipurpose conservation practices. Among the main functions are:

1. ***Reduction of runoff and soil erosion***: Agroforestry practices reduce transport of non-point source pollutants (e.g., sediment, chemicals) to waters (Fig. 10.2). Trees have inherent abilities to improve water infiltration, reduce runoff volume, and cleanup the polluted runoff and sediment. Specifically, eucalypts, poplars,

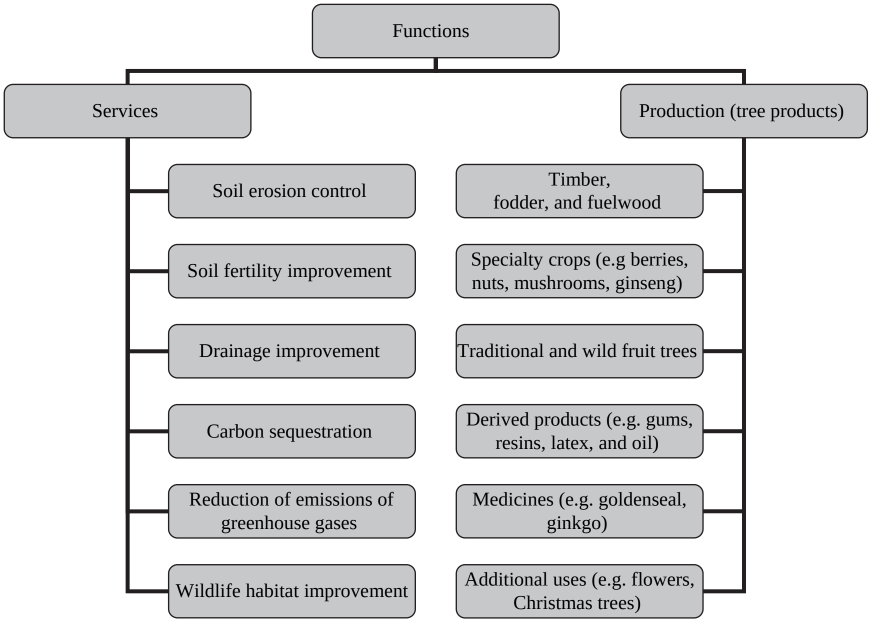

Fig. 10.2 Agroforestry systems have numerous functions

and willows can remediate environmental pollution while providing additional revenue from fuelwood and timber (Rockwood et al., 2004).

2. ***Land and wildlife habitat improvement***: Agroforestry practices improve soil fertility and productivity. These practices can restore degraded, marginal, and mined soils by improving the soil fertility and resilience. Planting trees and/or shrubs is also beneficial for wildlife habitat recovery/protection and diversification. Tree systems increase land value and improve landscape aesthetics. Agroforestry practice can improve water storage and use by balancing the components of the water balance (e.g., precipitation, runoff, soil evaporation, storage, drainage) by reducing losses due to runoff and evaporation and increasing transpiration.
3. ***Mitigation of global climate change***: Another crucial environmental function of agroforestry is the sequestration of C and reduction in net emissions of greenhouse gases to the atmosphere. Thus, agroforestry is a viable strategy to off-set CO_2 emissions by fossil fuel combustion. It offers the opportunity to sequester C within agricultural lands unlike traditional systems that separates agriculture from forestry. Well-managed agroforestry systems can meet societal needs by minimizing the use of non-renewable energy.
4. ***Food security***: Agroforestry comprises a set of innovative technologies to eliminate the hunger and reduce poverty by increasing crop and livestock production, especially in developing countries (Fig. 10.2). Trees and/or shrubs in agroforestry do not only improve the productivity of the companion crop and livestock systems but also provide tree-based or specialty products.

Agroforestry systems differ from traditional soil conservation practices (e.g. erosion control structures, grass strips) in that they are used for service and production. The numerous benefits can thus be partitioned into two main groups: (1) service and (2) production (Fig. 10.2). In addition to soil and water conservation, improvement of wildlife habitat, and mitigation of global climate change, agroforestry trees provide a number of specialty product and non-timber products such as food, fruit, fiber, and medicines, favoring the economics of growing trees. Additional income from multipurpose agroforestry systems compensates for crop losses that can occur due to weather inclemencies or other unexpected natural phenomena.

10.5.1 Magnitude of Soil Erosion Reduction

Soil erosion from sloping lands without agroforestry practices can be as high as 200 Mg ha^{-1} (Table 10.1). Introduction of agroforestry can reduce soil erosion by as much as 100 times in soils with steep slopes of up to 50%. Magnitude of reductions in soil erosion is region-specific and depends on differences in soil management, climate, and vegetation types. Hedgerows with trees and shrubs are as effective as traditional forest systems to reduce runoff and soil erosion. The effectiveness of agroforestry systems for controlling soil erosion is greater the more the system resembles the natural forestry in relation to litter abundance, spacing and height of trees. Agroforestry systems reduce runoff and soil erosion as much as does a no-till system.

Table 10.1 Effectiveness of agroforestry for reducing soil erosion

Country	Soil slope (%)	Soil erosion (Mg ha^{-1} yr^{-1})	
		With trees	Without trees
USA[1]	< 5	0.5	92
Jamaica[2]	24–32	0.5	1.4
USA[3]	2–5	0.2	0.3
Kenya[4]	20–40	6	11
India[5]	25–30	4	22
Philippines[6]	42	45	65
India[7]	4	12	39
Rwanda[8]	23–55	1–3	20–150
Peru[9]	15–20	0.2	53
Philippines[10]	14–21	< 5	100–200

[1]Pote et al. (2004), [2]McDonald et al. (2002), [3]Udawatta et al. (2002), [4]Angima et al. (2002), [5]Sharma et al. (2001), [6]Poudel et al. (2000), [7]Narain et al. (1997), [8]Roose and Ndayizigiye (1997), [9]Alegre and Cassel (1996), and [10]Paningbatan et al. (1995).

10.5.2 Agroforestry and Non-Point Source Pollution

The high reduction in runoff and soil erosion by agroforestry technology is positively correlated with the reduction in losses of sediment-bound and dissolved nutrients. Agroforestry practices such as riparian forest buffers remove significant quantities (>50%) of pollutants in runoff. Removal of pollutants in runoff of agroforestry trees is intrinsically related to the amount of water infiltration and runoff volume. Site-specific factors including type of pollutant, soil hydrology, tree species, and topography contribute to the variability in pollutant removal by trees. Pollutants filtered in surface runoff include nutrients, pesticides, animal wastes, and sediment leaving agricultural fields. Vertical and lateral flow of nutrients and pesticides into ground water are filtered and recycled by the tree root systems.

Water infiltration rates are normally high in soils under agroforestry practices. Thus, leaching of soluble chemicals with water infiltrating into the groundwater sources may be a concern. Leaching, which is the downward movement of soluble chemicals in percolating water within the soil profile, may be particularly a concern in humid or temperate regions. Tree species with shallow root system reduce leaching of pollutants.

Tree roots have, however, the ability to mitigate excessive leaching of nutrients through their "safety-net role", which refers to the ability of some tree species to recycle nutrients and enhance nutrient uptake while reducing chemical leaching (Allen et al., 2004). In steep sloping soils under alley cropping in Sri Lanka, *Calliandra* and *Flemingia* were identified as the most suitable species for reducing leaching losses of nutrients because of their slower decomposition rates and higher soil nutrient adsorption (De Costa and Atapattu, 2001). In claypan soils in the midwest USA, swamp white oak was one of the best species for reducing leaching of chemicals owing to its shallower and less concentrated root systems than other species (Udawatta et al., 2005).

10.6 Agroforestry and Factors of Soil Erosion

Agroforestry practices reduce soil erosion by altering:

1. rainfall and runoff erosivity,
2. soil erodibility,
3. land topography, and
4. surface cover

10.6.1 Rainfall and Runoff Erosivity

Agroforestry reduce soil erosion by *buffering raindrop impacts*, and *reducing runoff volume* and *velocity*. The canopy and floor litter of trees and/or shrubs reduces the rainfall and runoff erosivity. A dense canopy controls the rainfall erosivity by intercepting the falling raindrops and decreasing the raindrop erosive energy, whereas the dense floor litter layer beneath the forest canopy reduces soil detachment and splash and intercepts runoff flow and reduces its velocity, enhancing infiltration and sedimentation. The extensive tree root systems are pathways for rapid water infiltration and reduction in runoff volume. In fact, reduction in runoff through agroforestry trees is directly attributed to the increased water infiltration.

Soil hydrology of tree systems is different from that within cropped soils, and hence integration of trees with agriculture modifies the overall soil hydrology. Surface and subsurface runoff or interflow dynamics are altered with the agroforestry systems. Surface runoff practically accounts for the total runoff in agricultural soils, but within agroforestry systems such as alley crops or hedgerows, subsurface runoff and lateral flow/interflow are important components of total runoff.

10.6.2 Soil Erodibility

The combined effect of above- and below-ground biomass of trees makes agroforestry a valuable practice for maintaining and improving soil physical, chemical, and biological properties. Agroforestry practices improve the capacity of the soil to resist erosion by water and wind because they improve soil structural properties and drainage and increase soil organic matter content, macroporosity, water infiltration, and hydraulic conductivity.

The root system of trees holds soil in place and improves soil structural properties. The consistent decrease in bulk density across soils with agroforestry practices (Table 10.2) suggests improvements in total soil porosity and water transmission characteristics. For example, in Yurimaguas, Peru, the practice of planting trees decreased soil compaction as compared to land clearing with slash-and-burn in highly cultivated soils (Alegre and Cassel, 1996). In Nigeria, land clearing of trees for agricultural crops increased soil compaction (Lal, 1996). Compaction under agricultural crops increases runoff and soil erosion and reduces water transmission and retention because of decrease in macroporosity. Agroforestry practices increase water infiltration rate and saturated hydraulic conductivity. The depth of the forest floor and the coarse tree roots increase water infiltration and recharge. Infiltration in forest floors

Table 10.2 Impacts of agroforestry practices on soil bulk density and water infiltration

Soil	With trees	Without trees
	Bulk Density (Mg m^{-3})	
Ultisol[1]	1.01	1.12
Inceptisol[2]	1.24	1.48
Cambisols[3]	0.8	1.20
Alfisol[4]	0.66	1.41
	Water infiltration (mm h^{-1})	
Alfisol[4]	116	21
Lixisol/Alfisol[5]	135	44

[1]De Costa et al. (2005), [2]Vani and Bheemaiah (2003), [3]McDonald et al. (2002), [4]Lal (1996), and [5]Kiepe (1995).

is even greater than that in pasture grass due to differences in soil macroporosity and root system. Trees and shrubs have extensive roots which promote rapid water flow.

10.6.3 Terracing

Agroforestry practices reduce effective slope length and steepness. Wide, dense, and tall trees and/or shrubs planted across sloping soils such as those in contour hedgerows in alley cropping act like terraces for erosion control. Hedgerows develop terraces above them with time as a result of sediment deposition with soil eroded from the upper portions. A progression of evenly and closely spaced mini-terraces slowly develops under well-designed alley cropping systems (Fig. 10.3). This breakage in natural topographic structure has significant implications on soil hydrology and hence erosion. Other agroforestry practices such as riparian buffers and windbreaks also cause sediment build-up but their terracing effect is limited as these practices are normally established along the field perimeters or at wide intervals unlike alley cropping. The terracing effect of contour hedgerows is essential to: (1) reduce rill and prevent formation of gullies, and (2) increase the plant available water and groundwater recharge. Planting trees and/or shrubs along the existing mechanical terraces also improve the performance of both practices.

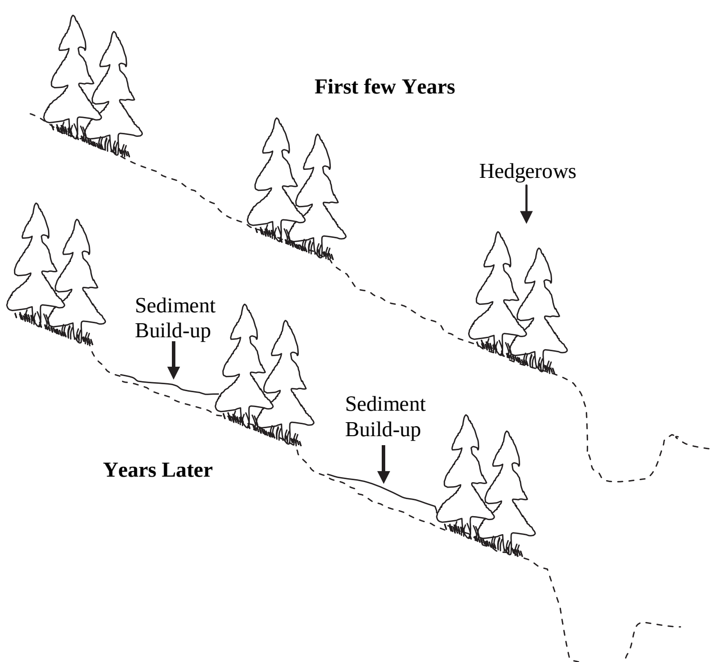

Fig. 10.3 Sediment deposition above hedgerows in alley cropping systems

10.6.4 Surface Cover

Dense and abundant canopy cover and surface litter are attributes by which agroforestry reduces soil erosion. Tree litter or mulch is the main soil erosion reducing factor. According to USLE, managing cover factor (C-factor) is critical to reducing soil erosion. Soil erosion increases linearly with decrease in canopy and mulch cover. A decrease in C-factor is associated with a linear decrease in the soil erosion rates. A bare soil has a C-factor of 1.0 whereas dense vegetation reduce this value to almost zero (<0.01). Hence, trees and/or shrubs in interaction with surface residues constitute a key biological treatment to minimize soil erosion. The surface litter protects the soil surface from raindrops impacts and improves soil structural stability and water infiltration. Surface cover also alters the soil-water relations (e.g. water retention, evaporation). It conserves soil water by reducing excessive evaporation. Prunings of contour hedgerows also reduce evaporation. Agroforestry that produces abundant and high quality litter is important to protecting soil from erosion and improving soil fertility.

10.7 Agroforestry and Land Reclamation

Agroforestry practices constitute an economical alternative to rehabilitate otherwise unproductive lands because these systems:

- reduce soil erosion and gullying,
- increase soil fertility and biological activity,
- improve soil properties and recycle nutrients ,
- increase efficiency of nutrient and water use, and
- improve drainage in poorly drained soils.

Selection of suitable species for each site-specific condition is essential for reclaiming lands. The recommended tree species include those with (Young, 1997):

- high above- and below ground biomass,
- deep and extensive root system
- high N-fixing capacity,
- no toxic substances,
- high capacity to grow in poorly drained and saline-sodic soils, and
- low competition and invasiveness rates.

Many trees grow in sloping and degraded/marginal lands and harsh climates. To counteract the shortage of arable lands, marginal or degraded systems must be reclaimed. Red gum, cottonwood, and Indian oak showed promise for reclaiming salt-affected lands in India (Singh et al., 1994), and Monterey pine and eucalyptus lowering groundwater table and reducing water salinity in western Australia, (Bari and Schofield, 1991). Agroforestry practices for reclaiming degraded soils are underutilized for reclaiming degraded ecosystems in spite of the beneficial aspects.

A direct benefit of agroforestry systems is the increase in soil fertility due to the addition of prunings (e.g. alley cropping) or residues from the trees (e.g., leaves, branches). This ecological function is critical to restoring degraded/marginal soils. Previously unfertile soils can be brought to production with the establishment of appropriate tree species to reduce rural poverty and hunger. Leguminous trees produce green manure as an alternative to costly inorganic fertilizers (Garrity, 2004). Agroforestry trees (e.g. leguminous) are able to fix N from the atmosphere and convert it to ammonia, which is returned to the soil in the form of litter fall or green manure. Decomposition of the tree residues or prunings increases the soil organic matter content and releases many essential nutrients, thereby improving soil fertility and microbial processes. Fertilizer trees and shrubs such as *sesbania, tephrosia, gliricidia,* and *wild wildflower* can provide 50–200 kg N ha^{-1} and increase crop yields by 2–3 times in eastern and southern African rural communities (Garrity, 2004). Conversion of natural forests to agriculture around the world has caused large losses in soil fertility. Agroforestry offers a sustainable alternative to monoculture row crops because of its potential to increase the soil organic matter content and improve productivity.

10.8 Agroforestry Plant Species

The number of tree and/or shrub species used in agroforestry is numerous (AFD, 2006; USDA-NAC, 2006). The selection of tree species for agroforestry is critical for a successful establishment and production of the different systems (Garrity and Mercado, 1994). Priorities for soil and water conservation, type of agroforestry practice, production and service, and preferences of the farmer/landowner dictate the choice of the appropriate species. Some of the decisive factors affecting the selection are:

1. ***Growth rate***. Trees should grow rapidly for maximum and prompt soil erosion control and production benefits.
2. ***Regrowth potential***. Ability of the trees to resist frequent cuttings and regenerate rapidly (coppicing ability) is important particularly in alley cropping.
3. ***Leaf and canopy characteristics***. Orientation and density of leaves play a major role in raindrop interception, filtering light or controlling shade for proper crop development. Leaf decomposition rate is another factor for nutrient recycling and supply in alley cropping, forest farming, and silvopasture.
4. ***Establishment***. Rapid establishment from seeds or cuttings and resistance to weeds as well as the ability to withstand pests, drought, waterlogging, and temperature fluctuations following establishment are essential qualities.
5. ***Root system***. Root distribution is important to stabilize the soil, recycle nutrients and minimize leaching of chemicals and achieve the “safety-net” role. In some ecosystems, trees with shallow root systems are preferable for reducing leaching.

6. ***Uses***. The most important criterion for selecting trees is the projected use of the plantations including timber, fruit, forage, green manure, and firewood production.

There are two systems of agroforestry for managing tree-crop interactions and competition for water, nutrients, and light: s*imultaneous* and *sequential* (Sanchez, 1995).

1. *Simultaneous* agroforestry is a system (e.g. alley cropping, contour and boundary hedges) in which trees and crops grow simultaneously on the same land. Tree-crop competition is high under simultaneous systems and low under sequential systems.
2. *Sequential* agroforestry is a system (e.g., shifting cultivation, improved fallows) in which trees and crops grow at different times. Unlike in simultaneous systems, demands for water, nutrients, and light do not overlap in sequential systems. Heightened demands for increased crop yield do not favor the use of sequential in some environments.

10.9 Alley Cropping

Alley cropping refers to the practice of planting agricultural or horticultural crops in widely spaced alleys between 1- and 5-m wide hedgerows of trees and/or shrubs (Kang et al., 1999) (Fig. 10.4). The alleys can be 10- to 25-m wide, depending upon the soil slope, width of the hedgerows, cropping system, tillage equipment, cultural operations, erosion hazard, and climate of the region. This agroforestry practice is also known as hedgerow intercropping. The hedgerows are established

Fig. 10.4 Soybean and black walnut alley cropping field (Courtesy USDA Agroforestry Forestry Center, Nebraska)

along the field contour with the annual or perennials crops paralleling the hedgerows for an effective soil erosion control. Hedgerows thus used are referred to as “contour hedgerows”. Alley cropping is one of the most widely studied agroforestry practices in tropical and subtropical regions, but it is relatively a new phenomenon in the USA, and currently it is mostly practiced in the midwest and southeastern states.

10.9.1 Benefits of Alley Cropping

Alley cropping offers many advantages over traditional agricultural systems including runoff and soil erosion control, improvement of soil properties, protection of wildlife habitat, and generation of additional income to small landholders.

1. ***Reduction in runoff water***: Hedgerows of high density act like terraces to reduce the runoff velocity and volume. In western Himalayan region of India, alley cropping in contour with *Leucaena* and eucalyptus hybrid reduced runoff by about 50% (Table 11.1). The effectiveness of alley cropping in reducing runoff from agricultural soils is high due to the positive effects of hedgerows on water infiltration. Periodic pruning and litter accumulation on the soil surface also contribute to improved water infiltration.
2. ***Reduction in soil erosion***: Hedgerows are especially useful in controlling erosion from sloping lands. In steep sloping soils in Indonesia, alley cropping reduced soil erosion by 64% while no-till reduced by 37% in croplands (Iijima et al., 2003. Alley cropping with no-till practices is more effective for erosion control than that with conventional tillage (Fig. 10.5). In Alabama, combination

Fig. 10.5 Orchard grass and black walnut alley cropping field (Courtesy USDA Agroforestry Forestry Center, Nebraska)

of mimosa, blackberry, and switchgrass in alley cropping has been used for soil erosion control (Shannon et al., 2002).

3. ***Development of natural terraces***: The terracing effect in alley cropping changes the thickness of the A horizon within the alleys. These changes in topsoil thickness create distinct zones within the alley with differing soil textural, structural, and hydrological characteristics. The lower alleys have higher values of saturated hydraulic conductivity and plant available water. Erosion at the upper alleys exposes layers of different texture and structure, altering water and air transmission characteristics.
4. ***Improvement in soil organic matter and soil structural properties***: Long-term and well-designed alley cropping systems generally enrich the soil organic matter content and result in favorable soil structural properties due to additions of prunings and tree residues The largest gains in soil organic matter content are observed under older than younger alley cropping systems. Soil compaction is lower while macroporosity is higher in alley cropping systems.
5. ***Increase in nutrient supply***: Alley cropping increases the supply and availability of nutrients for the crops and reduces nutrient losses. Combination of alley farming with other practices control nutrient depletion in low nutrient input agricultural systems for smallholders' farms. Periodic pruning of trees and/or shrubs in hedgerows in alley cropping provides C and nutrients to alley crops.
6. ***Diversification of farm products***: Alley cropping diversifies production and integrates two potentially different systems for increasing economic profits while conserving soil and water. By using this cropping system, it is possible to obtain two income sources from the same piece of land. The risks of annual crop production loss are offset by the diversified crop products from the trees. Establishing specialty crops or fast growing trees produce timber, fiber, and fruit, generating additional income source at different times of the year. Alley cropping is an economical means to overcome low agricultural production in tropical and subtropical regions of the world while controlling soil erosion. It is an alternative to slash-and-burn agriculture in tropical countries. In southern USA, alley cropping is integrated with hay, corn, cotton, watermelon, squash and ornamental or fruit shrubs such as blueberry and huckleberry (Workman et al., 2003).

10.9.2 Design and Management of Alley Cropping Systems

These systems are established to satisfy the goals of two neighboring and contrasting environments. The hedgerows alter the flux of water, air, and heat energy within themselves and the companion intercrops. The layout of the alley cropping systems is important to reduce competition among plant species, production, and land optimization. Well-planned alley cropping systems are environmentally sound system for controlling soil erosion and conservation water while providing a unique opportunity for generating income in a sustainable and methodical manner. Hedgerows can consist of single or mixed plant species planted in single or multiple rows (Fig. 10.6).

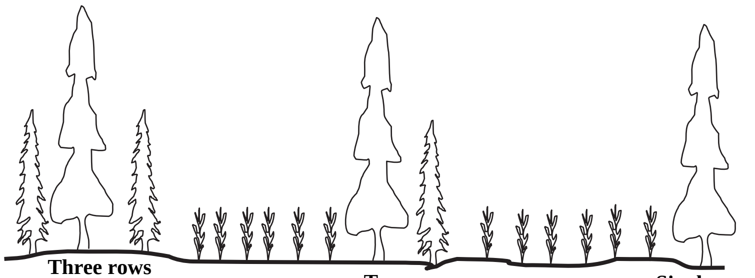

Fig. 10.6 Alley cropping system laid out on three-row, double-row, and single-row systems with crops in between rows

The following points should be considered when designing alley cropping systems (Garret and Buck, 1997):

- Rows of tree and/or shrubs must be spaced to accommodate the light, biological, thermal, and nutritional needs of the agricultural crops. Severe competition for water and nutrients reduces row crop production and/or tree growth in early stages of development.
- Proper selection of trees and crops, alley spacing, pruning, and timing of planting and harvest reduce competition.
- Mulching and use of cover crop must be components of the system.
- Corn, soybeans, wheat, sorghum, and potatoes require abundant light and should not be planted too close to the hedgerows.
- Shade tolerant species grow under or near trees.
- Spacing within hedgerows should be close to increase potential for runoff and soil erosion control.
- Spacing between the hedgerows ranges from 10 to 25 m, depending on the local and regional objectives and conditions.
- Expanded use of tree hedgerows for lumber and fruit production may require the establishment of wide within-row spacings.
- In the midwest USA, a 12 m spacing between hedgerows is an optimum width to reduce allelopathy and provide sufficient light for most crops up to about 10 yr following tree establishment. Some trees produce chemical compounds that inhibit the growth of neighboring plants, and this is known as allelopathy. Black walnut and pine bark and needles lower the soil pH to 4.5 or 3.5.
- A 25 m spacing between hedgerows would likely avoid shade overlapping up to about 20 yr. Longer periods of row crop production require wider spacing between hedgerows.
- Tree hedgerows should be established against the prevalent wind direction and east-west direction to provide enough light while reducing wind erosion.
- The tree rows should be oriented on an east-west direction to maximize the use of sunlight.

Example 1. How many rows of trees can one fit in one hectare of land if the spacing between rows is 12.5 m and the within-row spacing of trees is 2 m. Assume that the land is a square.

$$\text{Number of rows} = \frac{100}{12.5} = 8 \text{ rows}$$
$$\text{Number of trees} = \frac{100}{2} \times 8 = 400 \text{ trees}$$

Example 2. How many trees can be established in one hectare of a silvopastoral system if the tree spacing is 4 by 4 m and 9 by 9 m.

$$\text{First layout: Area per tree} = 4 \times 4 = 16\,\text{m}^2$$
$$\text{Second layout: Area per tree} = 9 \times 9 = 81\,\text{m}^2$$
$$\text{Number of trees} = \frac{10,000\,\text{m}^2}{16\,\text{m}^2} = 625$$
$$\text{Number of trees} = \frac{10,000\,\text{m}^2}{81\,\text{m}^2} = 123$$

10.10 Forest Farming

Forest farming is an intensive management where trees and other plant species are grown for the production of specialty crops. The specialty crops include wood products, fruit tress, food, decoratives, handicrafts, and medicine (USDA-NRCS, 1997). Forest farming is structured in multi-strata (multistorey) for the production of different crops at various layers. Trees and food crops occupy different canopy levels in the multistorey system. At the upper levels, soft- and hard-wood trees including shrubs are grown, while at the lower levels, shade-tolerant crops such as banana, coffee, pineapple and other specialty crops such as wildflowers, ginseng, herbs, ferns, and shiitake mushrooms are grown. Tall trees provide canopy cover for the optimum growth of lower level specialty crops. Pruning or thinning of upperstorey trees is necessary to facilitate sufficient light penetration for the lowerstorey crops. High-value specialty crops from the different levels make forest farming an attractive and profitable practice. Forest farming is a critical source of specialty crops and soft- and hard-wood, thereby easing the high burden on traditional forestry. It is important to the diversification and improvement of rural economy by generating short- (specialty crops) and long-term (trees) income sources with low investment although the labor can be intensive.

One of the intensive practices related to forest farming is the production of wood fiber for paper and strand board industries and is called *fiber farming*. Traditionally, wood fiber has been supplied by natural forest systems. Fiber farming, which emerged in 1990s, is an important practice to produce hardwood material in intensively managed forestry systems. Fiber farming entails cultural practices (e.g., soil preparation, fertilization, weed and pest control, irrigation) similar to those for agricultural crops (Yin et al., 1998). The fiber farming involves short rotations based on the best species suitable to a particular farm land, land owner, and fiber market demands. Thus, it differs from traditional forestry because of its careful and intensive management, resembling agricultural crops. It achieves greater fiber production per unit area and time than traditional forestry. Trees in fiber farming are harvested within 12- to 15-yr after establishment (Miller, 2004). Thus, time between planting and harvest is reduced by 75–85% relative to conventional forestry. The fiber farming is an innovative alternative fiber source as it is exclusively designed to produce wood fiber for papermaking.

Fiber farming is being practiced in several USA private lands and research centers particularly in the Pacific Northwest, Wisconsin, Minnesota, and recently in Michigan (Miller, 2004). One of the main farms for the production of fiber in the USA is the Boise's Washington Fiber Farm, Wallula, WA. Intensive forestry at this farm started in 1991, using hybrid cotton-woods, which are harvested after 6 yrs when the trees reach a height of 20 m. A similar site is at the Upper Peninsula Tree Improvement Center in Escanaba, MI where six tree taxa are grown on former agricultural fields to identify prominent taxa suitable for rapid production of hardwood fiber in the region (Figs. 10.7 and 10.8). Fiber farms use selected tree hybrids. Fiber farming provides an important option for farmers who are seeking economical alternatives to traditional crops.

Fig. 10.7 Several taxa of poplar hybrids starting their 10th growing season at the in Escanaba, Michigan, USA (Courtesy R.O. Miller, Upper Peninsula Tree Improvement Center, Michigan State Univ.)

Fig. 10.8 Two poplar varieties in their third growing season in Escanaba, Michigan, USA (Courtesy R. O Miller, Upper Peninsula Tree Improvement Center, Michigan State Univ.)

Soil and water conservation benefits under forest farming are similar to those under natural forests. It modifies the natural forest ecosystem but does not necessarily decrease the wind and water erosion control capacity compared to natural forest systems. Forest farming systems with a series of canopy layers within the multistorey structure reduce the erosive impact of raindrop and soil erosion more than monocultures provided that raindrops regain the erosive velocity under tall and monostorey tree plantations. Forest farming is established in natural forest and agricultural lands for intensive crop production. By growing alternative and ecologically sound trees, it is possible to thwart the deforestation and preserve the native forests. An optimum management of tree plantations demands resemblance of natural forestry. That is, forest farming should be more like natural forestry than agriculture.

1. ***Agricultural lands.*** Conversion of agricultural lands to forest farming is a viable alternative for soil and water conservation. It reduces soil erosion and improves soil functions, ecological biodiversity, and wildlife habitat. The vegetation under forest farming provides a more permanent and continuous surface cover than seasonal crops. Intensive tree plantations and matched with proper rotations function as well as natural forest systems for soil and water conservation purposes.
2. ***Marginal lands.*** Forest farming is also a strategy for reclaiming marginal, fallow, degraded or abandoned lands. The most viable forest farming system is establishing tree plantations on degraded or marginal agricultural lands rather than clearing native forest lands for the establishment of intensive plantations. Forest farming technology targeted to marginal agricultural lands is a win-win situation

for improving social and economic constraints by providing opportunities for additional income and employment.

3. ***Water conservation.*** Forest farming conserves water in that it uses water more efficiently (e.g. drip irrigation) than croplands which use center pivot and other large irrigation systems. It also benefits the water quality because it may use less fertilizers and pesticides than seasonal crops.

10.11 Silvopastoral System

Silvopastoral is an agroforestry system that integrates trees and shrubs with forage (pasture or hay) and livestock operations, and it is referred to as silvopastoral agroforestry. It is a multipurpose system where trees are deliberately and orderly combined with pasture and livestock to simultaneously enhance tree, forage, and livestock production. Within the integrated system, the three subsystems of production complement each other. The trees are planted in single or multiple rows with animals grazing in wide alleys between the rows. The trees are managed for dual propose: (1) wood production and (2) shelter and forage for farm animals. These systems provide economic returns from three sources (trees, forage, and livestock) at different times, creating diversified marketing and labor possibilities. Depending on the farmer's preference and ecosystem conditions, some silvopastoral systems emphasize one subsystem over the other. Silvopasture differ from traditional forestry because it is deliberately established and intensively managed for the optimization of benefits from tree-forage-animal systems. This multi-system creates microclimate zones, improves nutrient recycling, improves soil-water relations, enhances C sequestration, reduces emissions of greenhouse gases, and provides habitat and protection for wildlife.

10.11.1 Silvopastoral System and Soil Erosion

Silvopastoral agroforestry is an ideal system for soil and water conservation because trees and pasture prevent water and wind erosion. The dense root network under trees, shrubs, and grasses improves water infiltration, reduces runoff volume, and ameliorates transport of non-point source pollutants to downstream waters. Integrated systems of trees and/or shrubs with pasture can reduce runoff by 50 to 80%, sediment transport by 80%, and about 50% of total N and total P (Daniels and Gilliam, 1996). Less soil erosion occurs in silvopasture because the two predominant canopy levels (short and tall vegetation) intercept the raindrops and reduce runoff velocity. Tall trees act like windbreaks by filtering dust and reducing snow drifting from neighboring farms. Silvopastoral systems are especially suitable for degraded/reclaimed and sloping soil environments. In the USA, silvopastoral system is particularly popular in southeastern states.

10.11.2 Establishment and Management

When planning the establishment of a new silvopastoral system, the goals for tree, forage, and livestock crop production must be matched against the benefits of the systems for runoff and soil erosion control and improvement of water quality and wildlife habitat. Trees suitable for silvopastoral systems are those that:

- grow in marginal or degraded lands,
- are compatible with grass species,
- tolerate shade and occasional waterlogging, and
- permit intensive management and grazing and traffic.

Existing or native tree or grass species are the basis to establish the silvopastoral systems. Climate, landscape characteristics, vegetation, kind of livestock, and water source dictate the type of silvopastoral systems. Trees have to be evenly distributed to accommodate forage growth and support animal and farm equipment traffic. Establishing trees in rows along the field contour provides wide open spaces for livestock traffic and grazing while preventing gullying. Dense trees are pruned to enhance light penetration, enhance growth of trees and grasses, and reduce competition for water between trees and pasture. Thinning or pruning, fertilization, and use of leguminous trees and rotational systems are components of silvopastoral management. Grazing generally begins when the trees (two or three yr) are large enough to resist livestock traffic. Grazing under a controlled number of animals enhances nutrient cycling and reduces use of inorganic fertilizer. Well-managed grazing increases organic matter content and improves soil productivity.

10.12 Use of Computer Tools in Agroforestry

Planning, management, and development of optimum agroforestry systems require the use of advanced technological tools. Computers are becoming an integral component of decision making in the establishment and management of agroforestry systems in many regions of the world. At first, computer tools in agroforestry assist in collecting, storing, and organizing large quantities of laboratory and field data. Then, computers assist in the analyses and synthesis of data through computer-based support systems. Examples of such tools are the GIS and process-based and descriptive models.

10.12.1 Geographic Information Systems

The GIS is a computer-based system designed to store, analyze, and produce geographically referenced data or information. By bringing the spatial component into the computer database, GIS allows the assessment of the spatial relationships essential to selection of best locations for the establishment and management of agroforestry systems (Ellis et al., 2004). The GIS allows the conversion of databases to maps with specific information about geographic locations, tree species, soil types, landscape characteristics, and watercourses. The use of computers for agroforestry data management started at ICRAF or World Agroforestry Center in 1980's through the Agroforestry Systems Inventory Database (AFSI). The AFSI was replaced with the Multipurpose Tree and Shrub Database (MPTS), and then with the current database system, which is the Agroforestree Database (AFT). The AFT contains database with descriptions about tree species, ecological distribution, and management of more than 500 species. The AFT is specifically designed to assist with the selection of species for different agroforestry practices (AFD, 2006). Similar databases in other parts of the world have been developed for use in GIS and modeling. For example, the Florida Agroforestry Decision Support System (FADSS) in the USA is one of the GIS-based tools designed to explore agroforestry opportunities and identify potential tree species from a database of 500 trees (Ellis et al., 2000). Using FADSS, the farmer or landowner can pinpoint their land location and select tree species and type of agroforestry practices based on the production and management interest of the users.

Current trend is to integrate GIS, remote sensing, and mathematical models for a better understanding of complex and large-scale agroforestry systems. Robust GIS studies incorporate remote sensing technology to combine spatial data with satellite images and geographic locations. Global Positioning Systems (GPS) are GIS related tools to determine land elevations and coordinates with respect to references points. The GIS along remote sensing (e.g. Landsat images) provides maps and elevations while models predict effects of agroforestry management on the environment (e.g. soil, air and water quality) for a specific region. Scaling up the adoption of agroforestry requires the use of advanced computer-based tools as decision making tools. The cross combination of computer-based tools allows more complete characterization, better recommendation domains, and larger-scale measurement of agroforestry management impacts.

10.12.2 Models

Mathematical models in agroforestry systems are used to describe, understand, and predict the present and future functions of these systems in relation to their potential for soil erosion control, C sequestration, and alleviation of poverty. Models integrate a number of biological and physical characteristics in an attempt to explain the interactive and complex variations in agroforestry systems among ecoregions. Modeling agroforestry systems helps optimize design and management and explore new scenarios of land use while assisting agroforestry managers with decision making skills. A large number of models ranging process-based to empirical and descriptive approaches are available for agroforestry simulations. Some of the models are agroforestry-specific while others are not. Most models have been adapted from existing models in agriculture. Models including Century (Williams et al., 1984), Water, Nutrient and Light Capture in Agroforestry Systems (WaNuLCAS) (Van Noordwijk and Lusiana, 1998), CO_2FIX V.2 (Masera et al., 2003), and Agricultural Non-Point Source Pollution (AGNPS) and USLE (Kusumandari and Mitchell, 1997) are used in agroforestry research. WaNuLCAS has been used for modeling width, spacings, pruning regimes, safety-net of tree roots, and lateral tree-crop interactions of various types of agroforestry practices (e.g., alley cropping). WaNuLCAS has been used for identification that light and soil water were the yield limiting factors in trees and alley crops (Pinto et al., 2005). The CO_2FIX V.2 model is specifically designed to estimate the C sequestration potential of in traditional and intensively managed forest systems (Masera et al., 2003). The CO_2FIX V.2 is a comprehensive model that accounts for above- and below-ground biomass inputs including wood products, litter, humus, and wood recycling, and thus it has a wide applicability in temperate and tropical regions.

10.13 Challenges in Agroforestry Systems

Benefits of soil and water conservation of agroforestry practices are well recognized, but these systems are yet to be widely adopted. Adoption of agroforestry practices has been slow. Agroforestry systems must be made profitable. Examples of crop production increases, commercialization of tree-derived products, and provision of financial incentives might heighten farmer's interests in agroforestry practices. Agroforestry systems have potential to address the subsistence needs of rural communities, but a large scale adoption is needed to restore the vast marginal and degraded lands so as to effectively ensure food security, alleviate poverty, and sustain the environment. Development of regional and local programs that provide financial incentives to farmers who engage in agroforestry practices is a priority for a large-scale adoption of these systems to combat poverty.

Considerations in relation to social (e.g., demographic factors, land ownership, availability of markets, infrastructure), economic (e.g., financial incentives, economic benefits) and environmental (e.g., soil erosion, water quality, global climate change) constraints are essential to the success of agroforestry programs. Some of the technical obstacles that limit the rapid expansion of agroforestry for soil and water conservation are the lack of (Nath et al., 2005):

- knowledge concerning the design and management techniques,
- selection and domestication of potential tree species,
- supply or seeds of vegetative materials,
- large scale demonstration and commercialization of agroforestry tree products, and
- country- or region-specific programs for selection of species, management guidelines, and marketing.

There are also constraints in regards to:

1. ***Competition for water, light, and nutrients.*** Trees and shrubs (e.g., alley cropping) can compete with the companion crops for water and nutrients, resulting in

reduced crop yields. Tree roots that penetrate deep into the soil profile increase nutrient cycling and reduce competition for water in surface layers. Increases in soil fertility, C inputs, water use, uptake of leachable nutrients, and soil biomass by agroforestry practices over annual cropping systems may not always increase crop yield due to competition (Sudmeyer and Flugge, 2005).
2. ***Weed invasion.*** Weed invasion and tree shading reduce yields of sensitive crops.
3. ***Leaching of chemicals.*** Site-specific consideration to leaching of nutrients through tree rows is important. In some regions, deep-rooted trees cause significant leaching of nutrients.
4. ***Soil compaction.*** Frequent traffic for harvesting and cultural practices in intensively managed forest systems compact soil. Traffic-induced compaction during harvesting and site preparation affects growth of trees, offsetting any potential benefits on soil properties.
5. ***Soil properties.*** Impacts of different tree plantations on soil erosion and soil properties differ among tree species because of differences in biomass or litter input.
6. ***Crop yields.*** A balance between benefits to soil erosion control and crop production must be established to develop sound agroforestry ecosystems. Aside from the benefits for soil and water conservation and wildlife habitat improvement, agroforestry implications for maximizing tree, crop, and pasture yields are unclear. Components benefiting the optimum production of trees, crop, forage, and livestock may adversely affect soil erosion control, wildlife habitat, and C sequestration.

Summary

Agroforestry systems have potential benefits to soil and conservation and food production. They reduce soil erosion, improve soil properties and landscape aesthetics, mitigate the projected global climate change, and improve wildlife habitat and food security. Trees improve soil drainage, sequester soil organic C, reduce emissions of greenhouse gases, and provide specialty products while improving farm economy in poor regions. Agroforestry practices reduce soil erosion by changing the erosivity of rainfall and runoff, erodibility of soil, topography of land, and surface cover. These practices can also restore marginal and degraded lands by improving soil fertility, drainage, and soil properties. Legume trees fix N from the atmosphere while the surface litter increases soil organic matter content. Trees are selected based on their growth rate, regrowth potential, canopy characteristics, and root system.

Alley cropping is one of the agroforestry systems that is becoming popular for hedgerow intercropping particularly in tropical and subtropical regions. Forest farming is another practice where trees are grown for the production of specialty crops. An example of forest farming is fiber farming which refers to the production of wood fiber for paper and strand board industries. A multipurpose system that integrates trees and shrubs with forage and livestock operations is silvopastoral agroforestry where trees are managed not only for producing wood and forage but also for sheltering farm animals.

Agroforestry practices are practiced but not widely adopted. Improvements are needed in regards to design and management guidelines, monitoring of demonstrations projects, selection and domestication of potential tree species, supply or seeds of vegetative materials, large scale commercialization of agroforestry tree products, and region-specific programs for selection of species. Challenges in relation to competition for water and nutrients, soil compaction, and decreased crop yields must be also addressed.

Study Questions

1. A farmer in Ohio has divided his field into two plots: one for corn under moldboard plow and another for forest farming. The slope of the both fields is 6% that runs 50 m downhill to a stream. Tolerable soil loss (T) for this soil is 10 Mg ha^{-1} yr^{-1}. Estimate the annual soil loss from both systems using the USLE.
2. A farmer in Nigeria wishes to plant trees in parallel rows with a distance of 8 m between rows. How many trees can the farmer plant in a 10.5 ha field if the inter-row spacing between trees is 3 m?
3. Discuss the main differences among the various types of agroforestry systems.
4. What are the differences between silvopasture and pasturelands?
5. Discuss obstacles for the wide-scale establishment of agroforestry practices.
6. Describe the strategies required for the expansion of agroforestry.
7. How can computer models be used to manage trees?
8. Describe the main challenges in the expansion of agroforestry technology?
9. Discuss the management of agroforestry systems.
10. What is the erosion control effectiveness of agroforestry systems as compared to no-till alone?

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Chapter 11
Mechanical Structures and Engineering Techniques

Mechanical or engineering structures are designed to control runoff and soil erosion in fields where biological control practices alone are insufficient to reduce soil erosion to permissible levels. Because construction of engineering structures involves soil disturbance, change in landscape features, and some removal of land from production, biological practices such as residue mulching, no-till, reduced tillage, cover crops, riparian buffers, and grass filter strips must be the first choice for controlling soil erosion. Biological measures are also less expensive than engineering structures. Vegetative cover moderates erosion in a natural and ecological manner. Plants interact with the soil beneath in a mutual relationship while reducing soil erosion. Their roots increase soil shear strength and water infiltration and reduce detachment of soil particles. The canopy cover intercepts and changes the erosive raindrops into non-erosive streams of water throughfall. Dense stands retard runoff velocity and increase the infiltration opportunity time, thereby reducing runoff.

Mechanical structures are companion practices designed to improve the performance of biological conservation practices. On severely gullied terrains, biological practices must be supplemented by mechanical structures.

These structures are designed to:

- intercept and reduce runoff velocity,
- pond and store runoff water,
- convey runoff at non-erosive velocities,
- trap sediment and nutrients,
- promote formation of natural terraces over time,
- protect the land from erosion,
- improve water quality,
- enhance biodiversity of downstream water,
- prevent flooding of neighboring lands,
- reduce sedimentation of waterways, streams and rivers,
- create recreational opportunities, and
- provide diverse ecosystem services.

Nowadays, bioengineering techniques are increasingly being combined with traditional mechanical structures for controlling erosion. For example, geotextile fences

H. Blanco, R. Lal, *Principles of Soil Conservation and Management*,

made of plant materials are normally used in conjunction with engineering structures. A number of mechanical and engineering structures are available, some of which are permanent and others temporary. Permanent structures are built for long-term erosion control and are established for a long-term use. Such permanent measures include terraces, drop structures, spillways, culverts, gabions, ripraps, and ditches. In contrast, temporary measures include contour bunds, sand bags, silt fences, surface mats, and log barriers. The choice of mechanical measures depends on the severity of erosion, soil type, topography, and climate.

11.1 Types of Structures

11.1.1 Contour Bunds

Contour bunding consists of establishing earth or stone embankments of 1–2 m width on the field contour to reduce runoff velocity. These bunds divide the field in nearly parallel segments for reducing effective slope length. Contour bunds are appropriate for fields with permeable soils of gentle to moderate slopes. The earthen bunds are also called level terraces. A similar practice involves the use of earthen dams constructed across and above gullies to intercept runoff and permit the diversion or storage of runoff. Earthen berms are also constructed as temporary floodwater storage basins similar to ponds. The sediment accumulated above bunds or dams permits formation of natural terraces and enhances vegetation growth.

Stone bunds laid out perpendicular to the dominant slope are popular in mountainous regions for reducing soil loss from the integrated effects of water, wind, and tillage erosion. Well-maintained stone bunds develop terraces and decrease the slope length. In the highlands of Ethiopia, mean annual soil loss of 20 Mg ha^{-1} from cultivated lands was reduced to a negligible amount with the introduction of stone bunds (Gebremichael et al., 2005). Depending on the age, soil type, slope, and climate, stones lines reduce $>$60% of net soil losses (Nyssen et al., 2000). Stone structures are more resistant to erosion than grass strips and provide a total barrier against erosion. Stone bunds are used as alternative to grass strips in steep slopes for erosion control. Studies in the South American Andes have shown that establishment of stone bunds across hillsides significantly reduced soil loss on slopes between 33 and 78% (Rymshaw et al., 1997).

11.1.2 Silt Fences

Silt fences are filter barriers consisting of woven and unwoven geotextile fabric products (e.g., jute, polyethylene) anchored to vertical metal or wooden posts, which are laid out on the contour across the slope for reducing runoff velocity and filtering the sediment. These commercially available filters are installed at a range of parallel intervals on hillslopes, disturbed stockpiles, and along streams affected by rill and channelized flow. These porous filters are temporary measures until an effective erosion control practice or rapidly growing vegetation is established. By reducing runoff velocity, the fences allow sediment deposition on the upslope side while enhancing runoff water infiltration.

Because of its simplicity and relatively low cost, use of silt fences is preferred by construction industries, mining companies, and routine soil conservation programs for reducing off-site soil transport from disturbed sites. Silt fences intercept and release runoff at slower rates, changing the pattern of overland flow from rill to interrill flow. Fences provide environmental benefits by reducing transport of non-point source pollutants such as sediment and chemicals. Silt fences retain about 70–90% of total suspended solids in runoff (Barrett et al., 1998), but are not effective in controlling concentrated flow. Turbulent and concentrated runoff may inundate and overflow the fences (Barrett et al., 1998) (Fig. 11.1).

The flow rate (q) through silt fences is estimated using Eq. (11.1), a modified form of the orifice equation, assuming that water flows through the filter orifices in response to the upstream hydraulic head (Jiang et al., 1997).

$$q = CA\sqrt{2gH} \tag{11.1}$$

or

$$Q = Cmd_1 d_h \varphi(n)\sqrt{2gH} \tag{11.2}$$

Fig. 11.1 Sediment accumulation above silt fences can overtop them under concentrated flow erosion (Courtesy C.J. Gantzer, Univ. of Missouri, Columbia, MO)

where C is coefficient of discharge, A is area of the orifice, g is acceleration due to gravity, H is depth of ponded water upslope of the fence, m is number of openings in the horizontal direction, d_1 is opening size in the horizontal direction, d_h is opening size in the vertical direction, and $\varphi(n)$ is orifice coefficient equal to

$$\varphi(n) = \sum_{i=1}^{n} \sqrt{1 - \frac{2i - 1}{2n}} \tag{11.3}$$

where n is total number of openings in the vertical direction. The water flow and the sediment filtering capacity of geotextile fabric depend on its hydraulic conductivity.

11.1.3 Surface Mats

Surface mats or nets are temporary measures consisting of any permeable material made from natural or synthetic materials including jute, coir, paper, straw, nylon, and polyethylene. These permeable blankets are unrolled and pinned with hardwood pegs to disturbed and exposed soil surface to achieve a number of soil conservation and agronomic objectives including:

- protection of soil against erosion,
- establishment of vegetation,
- suppression of weeds,
- stabilization of disturbed sites,
- improvement of drainage,
- stabilization of streambanks,
- reduction of runoff transport capacity, and
- separation and filtration of sediment.

The surface mats provide an immediate control of soil erosion and are commonly used on sloping fields when permanent vegetation cover is being established. Natural geotextiles (e.g., coir, jute) are preferred over synthetic materials to reduce concerns of environmental pollution. Natural geotextiles resemble the soil surface and do not alter the solar radiation nor cause overheating of the soil surface compared to synthetic non-biodegradable polymer materials. Biodegradable mats are part of environmentally sound and economically accessible bioengineering techniques and are as effective as synthetic mats. When placed on top of seeded soil surface, surface mats:

- intercept, absorb, and dissipate the raindrop energy,
- reduce surface sealing and crusting,
- reduce soil detachment and splash erosion,
- improve water infiltration,
- press and hold the soil in place,
- keep the seeds in place,
- change the soil microclimate,

- reduce evaporation,
- increase soil water storage,
- increase soil organic matter (e.g., biodegradable geotextile mats),
- stabilize aggregates, and
- promote growth of vegetation.

As vegetation becomes established, the biodegradable surface mats become a composite solution to erosion (Rickson, 2006). Woven jute (500 g m^{-2}) and coir mat (400–900 g m^{-2}) have C-factor values <0.10 (Morgan, 2005). On sloping lands, synthetic or biodegradable geotextiles are often used in conjunction with crop residues overlaid with the geotextile. Biodegradable geotextiles degrade faster than synthetic geotextiles, and yet provide sufficient cover to soil until vegetated, and, in combination with the seeded vegetation, they increase soil's shear strength and cohesion, reduce detachment and soil erosion, increase soil organic matter and water content, and improve plant growth.

The effectiveness of surface mats for reducing erosion depends on the geotextile properties. Surface mats that have *rough texture* and high *water holding capacity* (e.g., jute) and high capacity to *pond water* can effectively reduce soil erosion (Rickson, 2006). Surface mats are particularly effective at reducing transport of sediment in runoff and improving the performance of other erosion control practices. Soils treated with polyacrylamide and overlaid with geotextile fabric reduce soil erosion rates to non-detectable levels as compared to soils treated with polyacrylamide alone (Blanco-Canqui et al., 2004). Geotextiles of coir are used to stabilize erodible steep slopes because coir geotextile has a high tensile strength due to its high lignin content (Vishnudas et al., 2006).

Mats have, however, numerous shortcomings to reducing concentrated flow erosion especially in steep slopes. The mesh openings must be large enough to allow an optimum growth of vegetation but small enough to reduce soil erosion. Mats provide only a temporary erosion control and require a good contact with the soil to reduce runoff and soil movement from underneath the mat.

11.1.4 Lining Measures

Erosion-affected channels, waterways, seashores, lakes, and ponds with slopes <25% can be lined with erosion resistant materials and structures such as rock ripraps, gabion mattresses, bricks or concrete pieces (Fig. 11.2) These revetments are designed to resist the continuous beating of water flow and strong waves. The ripraps consist of cobbles and small boulders spread along erodible land and water boundaries and channel bottoms to convey runoff at non-erosive velocities. Lining with rocks and stones is effective at reducing concentrated flow erosion and seepage-affected areas. The size of stones and extent of lined area depend on the extent of erosion. Ripraps and gabions are flexible, resilient to minor earth movement, and easy to build but are susceptible to failure under gullying and turbulent flow. Excessive seepage and sedimentation are some of the factors that limit performance of riprap revetment.

Fig. 11.2 Grade stabilization structures are established along drainageways to prevent gully erosion (Courtesy USDA-NRCS)

11.2 Farm Ponds

There are four types of ponds:

- Rain-fed ponds
- Groundwater-fed ponds
- Stream or spring-fed ponds
- Off-stream ponds fed with diverted water

11.2.1 Groundwater-fed Ponds

These ponds, also called dugout ponds, rely on shallow water tables and are constructed in low lying areas with slopes $<5\%$ where the prevailing water table is near the soil surface (about 1 m) (Fig. 11.3). These ponds are relatively inexpensive to build. Collecting groundwater in ponds may be beneficial to lower the water table in poorly drained soils. Such ponds with good quality or non-saline water provide a good habitat for fish.

11.2.2 Stream or Spring-fed Ponds

Stream ponds are established across a stream or below a spring to capture a fraction of the continuous or intermittent flow of water. For this purpose, dams are

Fig. 11.3 Rainfed pond used for livestock production (Courtesy USDA-NRCS)

constructed in depressions across streams without completely blocking the passing water. These ponds may require a periodic removal of sediment that may accumulate above the dam. Excessive sedimentation may reduce the pond depth and adversely affect the water quality. Design of stream or spring-fed ponds is more complicated than groundwater–fed ponds.

11.2.3 Off-stream Ponds

These impoundments are built off the stream channel from which a portion of water is diverted through a ditch or piped into the pond. A pump can be used to fill the pond with water. A weir structure is sometimes used to redirect water to the pond. Off-stream ponds increase the water temperature and affect the aquatic life in the stream channel. Improperly designed off-stream ponds cause water pollution by sedimentation. These ponds must be designed with filtration systems to prevent fish diversion with water into the pond. Use of buried feeder pipes equipped with proper inlet and outlet devices is recommended.

11.2.4 Rainfed Ponds

Rain-fed ponds are the most common type of ponds in arid and semiarid regions. The source of water for rain-fed ponds is surface runoff. These ponds are installed at the mid and lower-slope positions on steeply sloping watersheds to intercept and collect runoff by gravity. The amount of water stored in these ponds depends on the drainage area and annual amount of precipitation and runoff. While the first three types of ponds are supplied with water from relatively permanent sources of water, the rain-fed ponds rely on seasonal and intermittent overland drainage, and are common in arid and semiarid regions. In developing countries, the rain-fed farm ponds, also known as on-farm catchments, are typically small with storage capacity of about 1,000 m^3. Rainwater harvesting has been traditionally practiced in India, China, Middle East, and East Africa. Now, this practice is being adopted in other regions to minimize drought constraints (Sekar and Randhir, 2007).

Agriculture uses about 75% of the world's freshwater for irrigation (FAO, 2003). Availability of freshwater is waning due to the intensive agriculture, increase in urbanization, industrial uses, and waste of water resources. The water scarcity in arid regions is compounded by the unimodal rainy seasons and increased drought periods in drylands. Water is the single most important factor of crop production, and its scarcity reduces crop production and affects the livelihood of resource-poor farmers in arid and semiarid regions of the world. Thus, collecting runoff water and using it for irrigation or livestock during dry spells or seasons is a necessity in rain-fed dryland agriculture (Kuiper and Hudak, 2000). Water harvesting reduces soil erosion while allowing production of annual crops, increasing cropping intensity, and permitting crop diversification. Thus, rain-fed ponds accomplish a dual function of storing water for crops and livestock and recycling runoff which would otherwise exacerbate soil erosion hazard. Recycling runoff losses can increase crop production (e.g., vegetables) by as much as 40–200% (Bhati et al., 1997). If sufficient water is harvested, rain-fed ponds contribute to a successful completion of crop production despite irregularities in rainfall. Conservation practices such as residue mulching, ridge tillage, no-till, and permanent vegetative cover also conserve rainwater in the root zone.

11.2.5 Design and Installation of Ponds

Proper design of ponds is important to reducing runoff, increasing water storage, decreasing water losses through seepage and overflow, and increasing water availability to crops. Suitable sites must be selected based on rainfall patterns, soil hydrology, and topographic features. Models and GIS can be used as decision support systems for designing ponds at large scale (Sekar and Randhir, 2007). The GIS-based models predict rainwater potential, display index maps of rainfall, runoff, and water harvest, and permit the identification of suitable spots for siting ponds (Senay and Verdin, 2004). Modeling of ponds consists of processing and integrating information from a database of climate, soil, land use, and landscape characteristics. Factors which influence pond design include:

- Watershed size
- Soil textural characteristics

- Soil hydrology
- Slope gradient
- Tillage and cropping systems
- Rainfall patterns
- Runoff amount
- Interflow and seepage problems

Water seepage is a major source of water loss from ponds. Thus, ponds must be made nearly impermeable to reduce seepage losses. Knowledge of soil profile hydraulic conductivity, infiltration capacity, drainage classes, lateral flow, and groundwater recharge is important to siting a pond. Compaction and lining with impermeable materials such as clay or plastic sheeting are useful strategies to reduce seepage in small ponds. Check dams and ripraps are used to protect ponds from overflow and sedimentation.

The volume of runoff available for harvesting is computed using Eq. (2.29). The pond depth is an integrated effect of numerous factors including evapotranspiration, cost of construction, safety, slope gradient, and land availability. For example, shallower ponds (about 1 m deep) may lose significant amounts of water through evaporation, but are relatively less expensive. The depth of pond (*D*), the drainage or watershed area (ha) required (*WA*) per pond, and the number of ponds (*NP*) for collecting about 1000 m^3 can be estimated using Eq. (11.4) (Senay and Verdin, 2004):

$$D = 1 + Evaporation - Rainfall + Seepage \tag{11.4}$$

The amount of evaporation (m), rainfall (m), and seepage (m) refers to annual averages. The *D* is set to 1 when it is negative as in Eq. (11.5)

$$WA = D \times \left(\frac{1000}{RF}\right) \times 0.0001 \tag{11.5}$$

where *RF* is average runoff depth available (m). The number of ponds is computed as per Eq. (11.6)

$$NP = \frac{TWA}{WA} \tag{11.6}$$

where *TWA* is total watershed area (m^2). The *NP* for large geographic areas is computed by increasing *TWA*. The number of ponds depends on the population density and specific use of water.

Example 1. Design a pond to supply water to 50 milk cows, 100 sheep, 0.5 ha of vegetables, and a family size of 5 in an arid region. Assume that losses of water in the pond by evaporation and seepage are about 50%. The average annual precipitation is 400 mm, and the runoff curve number is 70.

The water requirement is calculated using the data from Table 11.1:

Table 11.1 Approximate water requirement in the USA (Midwest Plan Service, 1987)

Type of water use	L d^{-1}
Human use per person	220
Milk cow	140
Dry cow	90
Calves	30
Bull, horse, mule, and donkeys	40
Swine	15
Sheep	9
Chicken: laying and non-laying hens (100 head) (moderate temperature)	20–27
Chicken: laying hens (100 head) ($\geq 32^\circ$ C)	32
Turkey (100 head): 30–70 Kg of weight	30–70
Turkey (100 head): 100–220 Kg of weight	100–220
Turkey (100 head): 230–460 Kg of weight	230–460
Irrigation in arid regions	7500 m^3
Irrigation in humid and temperate regions	2500 m^3

Water use	Animal unit	Total requirement	
	L d^{-1}	L d^{-1}	m^3 yr^{-1}
50 cows	140	7000	2555
100 sheep	9	900	329
5 persons	220	1100	401
0.5 ha		7500	3750
Total		16500	7035

The annual requirement of water (WR) is 7035 m^3, which must be adjusted for the losses as

$$WR = \frac{7035}{1 - 0.50} = 14070\,\mathrm{m}^3$$

The pond must have a capacity of about 13500 m^3 to satisfy the annual local requirements. The amount of runoff is estimated as

$$S = \frac{25400}{CN} - 254 = \frac{25400}{70} - 254 = 108.9$$

$$Q = \frac{(I - 0.2S)^2}{I + 0.8S} = \frac{(500 - 0.2 \times 108.9)^2}{500 + 0.8 \times 108.9} = \frac{228694}{587} = 390\,\mathrm{mm}$$

$$\text{Drainage area required} = \frac{WR}{Q} = \frac{14070}{0.39} = 36077\,\mathrm{m}^2 = 3.61\,\mathrm{ha}$$

11.3 Terraces

Terraces are earthen embankments established across the dominant slope partitioning the field in uniform and parallel segments. These structures are often combined with channels to redirect runoff to a main outlet at reduced velocities. Terraces have been used since the dawn of agriculture in many parts of the world (e.g., China, the Himalayas, Peru, Bolivia) for growing crops in hillsides (Fig. 11.4). In mountainous regions, the common terraces are narrow and most of them are constructed by hand. The advance in agricultural mechanization particularly in developed countries has made possible the construction of terraces and outlets across large fields. In the USA, broad-based, graded, and parallel tile outlet terraces are common whereas bench terraces are prevalent in other parts of the world. Graded terraces are slightly inclined along the dominant slope and accompanied by waterways and outlets to dispose off the runoff water. The tile outlet terraces collect, pond, and convey runoff in underground tile lines. Terracing is similar to contouring and decreases the P-value of the USLE by half as compared to strip cropping. The LS factor is reduced considerably by terraces because it is defined by terrace spacing rather than by the field slope.

Terraces provide the greatest benefit to soil and water conservation when used in conjunction with: (1) proper cropping and tillage systems such as no-till, reduced tillage, residue mulching, crop rotation, contour strip cropping, and soil conservation buffers, and (2) other soil conservation structures such as grassed waterways, drainage channels, underground outlets, sediment control basins, drop structures, and gabions.

Fig. 11.4 Hillside terraces are strategies to reduce soil erosion and stabilize landscapes (Courtesy USDA-NRCS)

11.4 Functions of Terraces

Terraces are constructed to achieve numerous functions. The main goal is to conserve soil and water. By decreasing the slope length, terraces not only allow the farming of steep slopes while reducing soil erosion risks. In mountainous areas, farming of hillsides would be nearly impossible without terraces. The loosening of soil during terrace construction increases the topsoil depth and facilitates crop establishment. In dry regions, terraces increase plant available water storage and groundwater recharge.

Terraces are important to:

- slow runoff velocity and reduce formation of peak runoff rates,
- reduce the slope length of the hillsides by splitting the field into narrow bands,
- reduce soil erosion and concentrated runoff,
- promote soil water storage by slowing and retaining runoff and promoting infiltration,
- reduce wind erosion by increasing soil water content and increasing surface roughness,
- facilitate surface irrigation in relatively level soils and increasing crop production, and
- improve water quality by allowing removal of sediment and chemicals from runoff.

11.5 Types of Terraces

There is no unique classification system of terraces. The American Society of Agricultural Engineers (ASAE) (2003) has grouped terraces based on *alignment, cross section, grade,* and *outlet* (Fig. 11.5). In general, there are four main types of terraces: *broad-base, narrow-base, bench*, and *steep backslope terraces*. Most terraces possess four sections: *wide segment* (between ridges), *cut-*, *front-*, and *back-slope segments*. Terraces are termed continuous if they cover large areas of the field and discontinuous if they are small and localized (e.g, orchard terraces, individual basins). Some terraces are transitional and can be removed when necessary. Choice of terraces depends on the dominant use (e.g., erosion control, water conservation). When water conservation is a major concern, broad-base and drainage terraces are preferable to absorb and store rainwater. In soils prone to erosion, however, bench terraces or traditional terraces are appropriate. Terraces are most suited to terrains with slopes $>5\%$. On sloping lands, terraces are primarily installed to grow crops without causing excessive soil erosion. In industrialized countries with large mechanized farms and relatively gentle slopes, broad-base terraces are the preferred type. Drainage type terraces are used in regions with high precipitation and poorly drained soils whereas absorption type terraces are preferred in regions with limited precipitation and permeable soils.

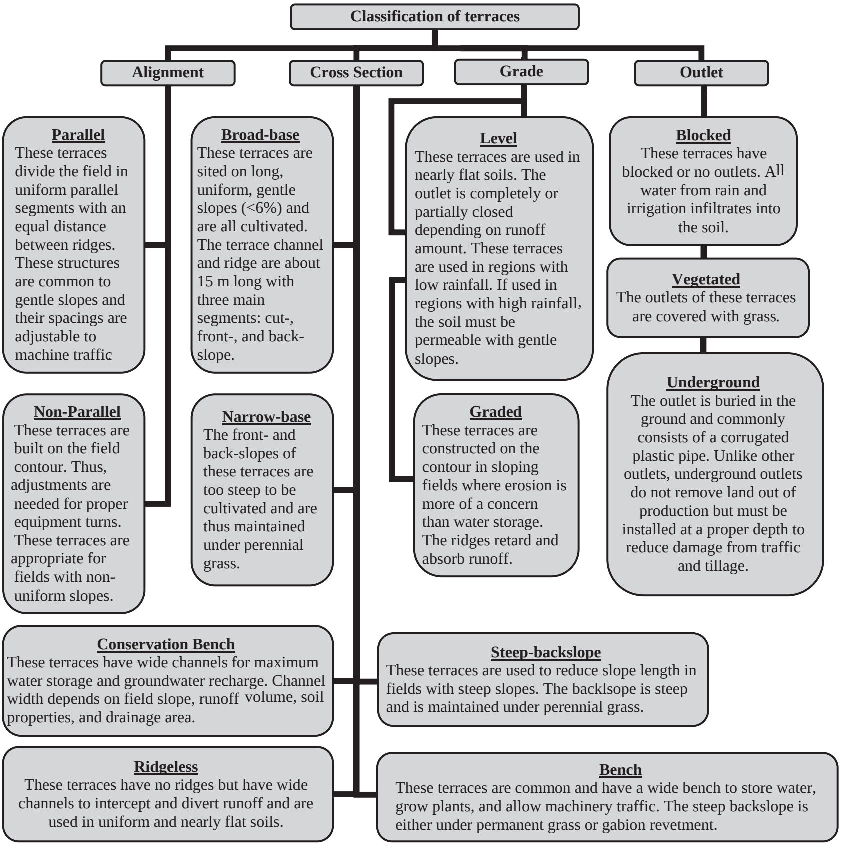

Fig. 11.5 Types of terraces (After ASAE, 2003)

Among the factors that determine the design and layout of terraces are:

- field topography,
- climate (rainfall, wind),
- soil type,
- tillage and cropping system,
- cost of construction,
- accessibility to heavy equipment,
- population density, and
- land ownership.

Terraces are designed to modify the original topography. The bed width of the terraces becomes narrower with increase in slope gradient and length. Narrow beds are more susceptible to high surface runoff velocity and soil erosion. Proper design of terraces is critical with increase in slope gradient. Rills start in the upper portions of the terrace and develop into gullies cutting through the lower terraces, especially when terraces remain bare and are improperly designed.

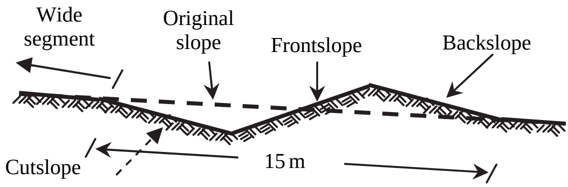

Fig. 11.6 Cross sectional view of lower portion of a broad-base terrace. The broad-base consists of lower and upper section. The lower section confines the channel and ridge (about 15 m wide) while the upper section confines the wide segment (about 30 m wide) (After ASAE, 2003)

Broad-base terraces. These terraces are used in long and uniform fields with slopes <5% and thus all the sections of the terraces are farmed (Fig. 11.6). They are also known as channel terraces because the channels are all cultivated and are common in regions with flat and abundant land, and where heavy equipment is available. Channels are gently graded to outlets for runoff disposal. These terraces are appropriate for regions where both soil erosion and drainage are required. Sheet and rill erosion between terraces are higher on broad base than on narrow base or grass backslope terraces in sloping fields.

Narrow-base terraces. These terraces are used in shallow and sloping lands unlike broad-base terraces and have a steep narrow ridge. The ridges are not farmed but are maintained and managed with permanent vegetation cover (Fig. 11.7). These terraces cause lesser soil disturbance than other types of terraces. Narrow terraces are common in regions with limited land and with steep slopes,

Bench terraces. These terraces are widely used throughout the world, particularly in hilly terrains. Bench terraces established on slopes >10% have steep backslopes. The width of the bench and the height of steps are variable depending on the field slope (Figs. 11.8 and 11.9). A width of 7.5 m is used in moderately sloping lands. Bench terraces are a series of strips constructed across the slope at equidistant vertical intervals and separated by steep banks of stones and grassed revetments.

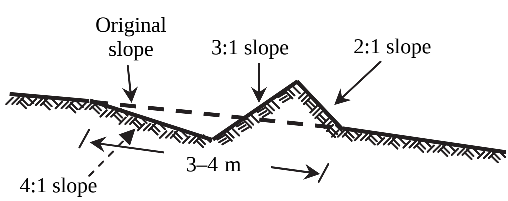

Fig. 11.7 Cross sectional view of lower portion of a narrow-base terrace. The ridge is steeper than that in broad-base terrace (After ASAE, 2003)

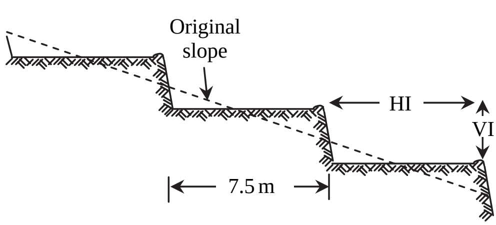

Fig. 11.8 Cross sectional view of a bench terrace. The HI signifies the horizontal interval and VI the vertical interval

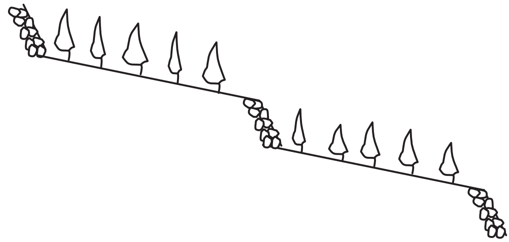

Fig. 11.9 Cross sectional view of a bench-type terrace with stone walls in the backslope

There are two types of bench terraces: *conservation* and *upland terraces*. *Conservation terraces* are also referred to as irrigation or level bench terraces and are used for flood irrigation (e.g., rice) and water storage. *Upland terraces* are used for rain-fed and irrigated crops and are gently sloped outwardly for allowing drainage. Bench terraces are more suited to large fields with mechanized agriculture under high-value crops.

Fanya juu terraces are used as alternative to bench terraces in some parts of the world particularly in regions with rugged topography (e.g., East Africa). *Fanya juu* terraces consist of embankments and ditches built by digging a trench on the field contour and throwing the excavated soil uphill (Fig. 11.10A). The embankment must be seeded to grass for proper stabilization. The trench is about 50 cm wide and 50 cm deep, resulting in about 50 cm high by 100 cm embankment. The distance between these terraces is a function of the field slope and can be 20–30 m wide in gently sloping lands and about 5 m in fields with steep slopes. The embankments accumulate sediment above and create natural terraces over time (Fig. 11.10B). In Tanzania, Tenge et al. (2005) reported that soil loss was reduced from 25–15 Mg ha^{-1} by grass strips, to 6 Mg ha^{-1} by bench terraces, and to 3 Mg ha^{-1} by *fanya juu* in soils with steep slopes up to 60%. The same study showed that corn grain yield was increased

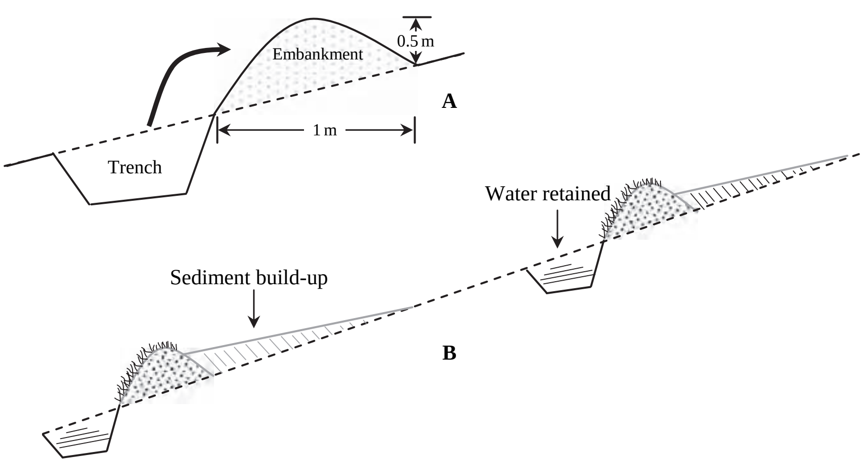

Fig. 11.10 A *fanya juu* terrace built from earth embankments (**A**) at construction and (**B**) after various years (Not to scale)

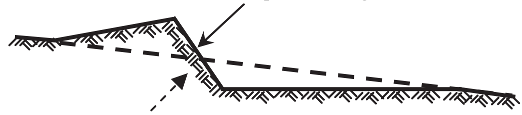

Fig. 11.11 Cross sectional view of a steep-backslope terrace (After ASAE, 2003)

by 30% for grass strips, 100% for bench terraces, and 34% for *fanya juu*. In the highlands of Ethiopia, soil loss was reduced from an average of 39.5–1.8 Mg ha^{-1} by establishing *fanya juu* terraces (Herweg and Ludi, 1999).

Steep backslope terraces. These terraces are all farmed except the backslope section which is permanently vegetated (Fig. 11.11). The front slope is wide for equipment maneuvering and is not as steep as in the narrow-base terraces. These terraces are used for slopes $<$15%.

Individual terraces. These terraces are small and round and are commonly used for planting individual plants (Fig. 11.12). They are appropriate for growing tree crops or other perennials.

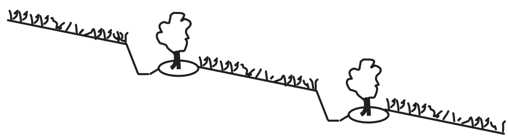

Fig. 11.12 Side view of intermittent or individual terraces for tree

11.6 Design of Terraces

Terraces must be designed to contain runoff for rainfall events with a return period of at least 10 yr. Terraces must be designed taking into consideration the soil slope gradient, risks of within-terrace erosion, soil properties, field equipment width, and cropping systems. They must have proper ridge height and well-designed outlet channels to convey water and reduce risks of overtopping. The design consists in determining the horizontal interval (HI) and vertical interval (VI) (Fig. 11.13). A number of empirical equations are available to determine the VI and HI of terraces (Morgan, 2005).

The cut and fill for constructing terraces are determined as follows:

$$c + f = h + sW \tag{11.9}$$

For balanced cross section (Fig. 11.13):

$$c = f = 2c = 2f$$

$$2f = h + sW \quad \text{or} \quad 2c = h + sW$$

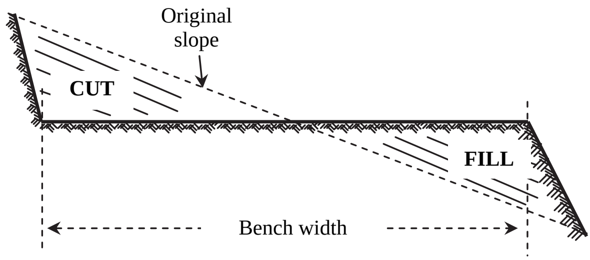

Fig. 11.13 Cross-section of a level bench terrace showing the cut and fill sections

Thus,

$$c = f = \frac{h + sW}{2} \tag{11.10}$$

where c is cut (m), f is fill (m), h is depth (m) of channel including freeboard, s is field slope, and W is width of the side slope, which is equal to the equipment width (Fig. 11.14). Terrace height is equal to the channel depth plus freeboard.

In the USA, terraces are designed based on the following relationships (ASAE, 2003):

$$VI = X \times S + Y \tag{11.7}$$

$$HI = \frac{(X \times S + Y)\,100}{S} \tag{11.8}$$

where X is based on the geographical location (Fig. 11.15) and Y is soil condition. The *VI* is difference in height between two adjacent or succeeding terraces. The *HI* is actual horizontal distance between terraces and not the distance measured over the land surface. The number of terraces in a long field is determined by dividing the width of the field by *HI*. The values of Y depend on the soil erodibility and tillage and cropping systems and ranges between 0.3 and 1.2 with commonly used values of 0.3, 0.6, 0.75, 0.9, and 1.2. A value of 0.3 is used for highly erodible soils with little or no crop residue/vegetative cover during intense rainfall periods. In contrast, the value of 1.2 is used for soils with low erodibility and mulched with

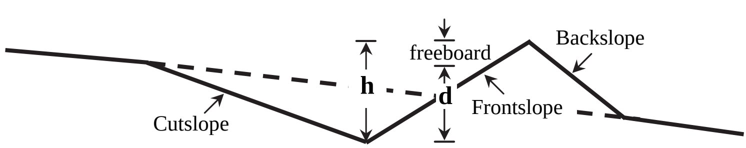

Fig. 11.14 Cross section of a terrace showing the terrace height (h), depth of channel (d), and freeboard

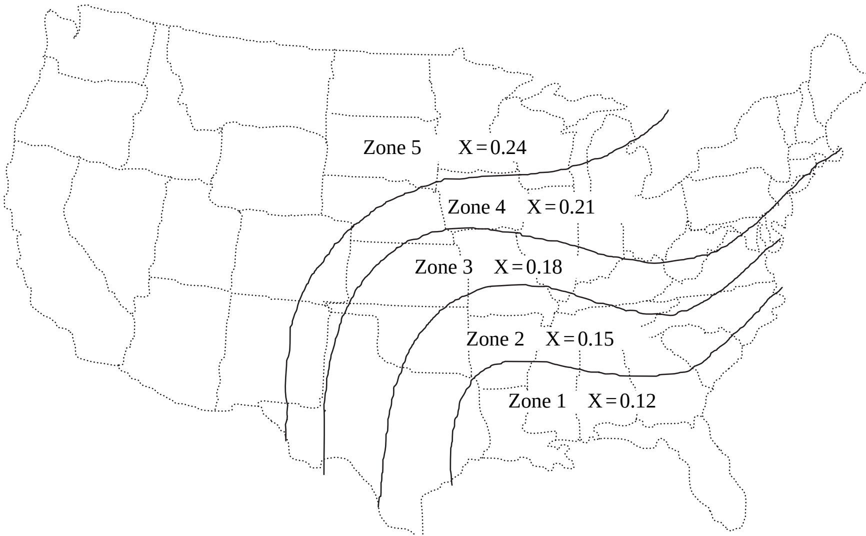

Fig. 11.15 Values for geographical location (X) in Eq. (11.7) for the USA (After ASAE, 2003)

at least 3.3 Mg ha^{-1} of crop residue. A Y value of 0.75 is used for soils with some favorable and other unfavorable factors.

Example 2. Compute the terrace spacing for a soil with moderate erodibility and 3% slope (Y = 0.75) in Minnesota. Determine the even multiple of turns between terraces if the width of the field equipment is 7.5 m.

From Fig. 11.15, X = 0.24

$$VI = X \times S + Y = 0.24 \times 3 + 0.75 = 1.47\,\text{m}$$

$$HI = \frac{(X \times S + Y)\,100}{S} = \frac{(0.24 \times 3 + 0.75) \times 100}{3} = 49\,\text{m}$$

The *HI* must allow an even turn of field machinery.

$$\#\text{turns} = \frac{49\,\text{m}}{7.5\,\text{m}} = 6.53$$

The *HI* can be adjusted to 45 m $\left(\frac{45\,\text{m}}{7.5\,\text{m}} = 6\right)$ so that the field equipment could make an even turn of 6 between terraces.

Example 3. Compute the cut and fill and front and back sideslopes to construct a terrace on a sloping field (5% slope) with a balanced cross section. The terrace width is 8 m and the channel flow depth is 0.5 m. The freeboard is 0.15 m.

Terrace height = h = d + freeboard = 0.5 m + 0.15 m = 0.65 m.
Required depth of cut and fill:

$$c = f = \frac{h + sW}{2} = \frac{0.65\,\mathrm{m} + 0.05 \times 8\,\mathrm{m}}{2} = 0.525\,\mathrm{m}$$

$$Front\ sideslope = \frac{Run}{Rise} = \frac{8\,m}{0.525\,m} = 15.24 \quad or \quad 15:1$$

$$Backslope = \frac{Slopewidth}{(h + s \times W)} = \frac{W}{(h + s \times W)}$$
$$= \frac{8\,m}{(0.525 + 0.05 \times 8)\ m} = 8.65 \quad or \quad 9:1$$

The length of terraces is determined by the size and slope of the field and soil erodibility (e.g., texture, permeability). Bench terraces are about 100 m long in regions with high rates of rainfall and longer in arid and semiarid regions (FAO, 1986). Bench terraces are suited to fields with slopes >10%. The width of these terraces depends on the soil depth, cropping and tillage system, and use of farm equipment. Cut and fill increase with an increase in terrace width, concomitantly increasing the cost of construction. The channel grade for conveying runoff must be between 0.5 and 1%. Use of heavy equipment (e.g., bulldozers, blade grader, moldboard plow) is used for terrace construction in large fields with slopes of up to 35%. On steeper slopes, terraces are manually constructed.

The FAO suggests the following relationships for designing broad- and narrow-base terraces (FAO, 1986):

In humid regions with erodible soils and narrow terraces:

$$VI = \frac{S + 4}{10} \tag{10.11}$$

In semiarid regions with normal soils:

$$VI = \frac{S + 6}{10} \tag{10.12}$$

$$HI = \frac{VI}{S} \times 100 \tag{10.13}$$

or

$$HI = \frac{VI}{\tan 0} \tag{10.14}$$

According to FAO (1986), the depth of cut (C_d) which is equal to depth of fill is estimated using Eq. (10.15) once the runoff depth or volume is determined:

$$Cd = \frac{h + sW}{2} \tag{10.15}$$

where h is depth of channel including freeboard (cm), s is original field slope (%), and W is width (cm) of side slope which depends on the width of the farm equipment in mechanized agriculture. A minimum of 4 m of side slope width is recommended in mechanized cultivation. The length of broad-base terraces must be <350 m for one flow direction, and the recommended width is 8–15 m for mechanized agriculture and 3–4 m for narrow-base terraces with a channel grade <0.5% (FAO, 1986).

For bench or level terraces, the design relationships according to FAO (1986) are:

$$H_r = VI + DH \tag{10.16}$$

where H_r is height of riser (cm) and DH is dike height equal to 15 or 20 cm.

$$VI = \frac{S \times W_b}{100 - (S \times U)} \tag{10.17}$$

where W_b is width of the bench, U is slope of the riser, which is 1 (1:1) for terraces constructed with mechanized equipment, 0.75 (0.75:1) for hand-made terraces with compacted earth, and 0.5 (0.5:1) for hand-made terraces with rocks (FAO, 1986).

The cut depth for bench terraces is estimated as

$$C_d = \frac{W_b}{2} \tan\theta \tag{10.18}$$

where $\tan\theta$ is the tangent of the slope angle.

Example 4. Estimate the vertical interval of a bench terrace with a width of 5 m on a field with 25% slope for hand-made terraces with rocks.

$$VI = \frac{S \times W_b}{100 - (S \times U)} = \frac{25 \times 5}{100 - (25 \times 0.5)} = 1.43\,\text{m}$$

11.7 Management and Maintenance of Terraces

Terraces are effective in reducing soil erosion and allowing crop production in erosion-affected soils. These structures must be accompanied by regular maintenance and monitoring of performance and functionality. One of the shortcomings of sloping terraces in mountainous regions is the systematic downslope translocation of soil by tillage and water erosion. The translocated material accumulates in the

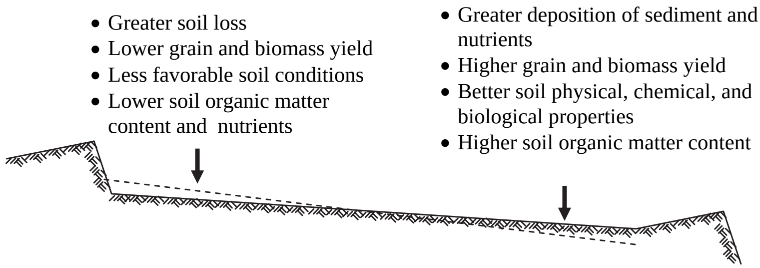

Fig. 11.16 Changes in crop yields and soil properties within a bench terrace in sloping soils

lower boundaries of the benches (Siriri et al., 2005). Excessive translocation of soil materials from the upper to the lower boundaries of the terrace causes variability in crop yields and soil properties with time (Fig. 11.16). Systematic and preferential removal of soil from the upper sections expose subsurface horizons, change surface soil properties, and thus decrease crop yields in the upper sections (Fig. 11.17). While the lower crop yields in upper sections may partly be compensated by the higher yields in lower positions, excessive translocation of soil may affect the stability of terraces and eventually reduce yields. Soil properties and crop yields change gradually with distance from the upper to the lower terrace riser. On steep soils in Uganda, soil organic C content, clay content, and soil bulk density were lower and hydraulic conductivity and crop yields were higher in the lower positions of the bench terraces (Siriri et al., 2005). Similarly, on terraced steep hillslopes in the Chinese Loess Plateau, the upper portion of the terraces had lower C content and higher bulk density whereas the opposite was true in the lower portions of each terrace due to the change in slope gradient (Li and Lindstrom, 2001).

Grain (Mg ha^{-1})	0.29	0.57	1.11	1.74	3.33
Soil organic C (g kg^{-1})	21	23	25	26	28
Bulk density (Mg m^{-3})	1.41	1.34	1.26	1.20	1.18
Saturated hydraulic conductivity (mm h^{-1})	16	23	38	66	73

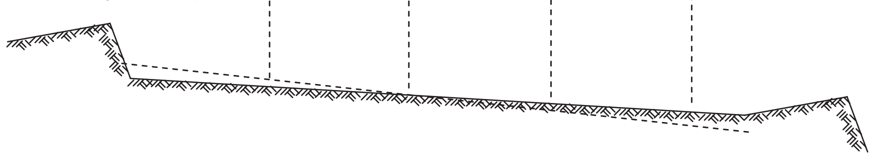

Fig. 11.17 Variations in sorghum yield and soil properties along a sloping bench terrace in Uganda (After Siriri et al., 2005)

Some suggested considerations to maintain terraces include:

- Height and stability of the ridges must be inspected and repaired after heavy rains.
- Plowing up ridges may be required to clean out the deposited sediment above the ridges and maintain an adequate ridge height for reducing overtopping.
- Any plow is sufficient for small repairs but heavier equipment (e.g., bulldozer or scraper) may be required for reconstructing large terraces.
- The topsoil removed during terrace construction must be replenished with proper residue mulching, manuring, and fertilization.
- Excessive erosion from upper boundaries must be controlled to reduce sediment deposition and burial of drainage pipes and maintain high water storage capacity.
- Hauling back the eroded sediment from the lower to the upper positions may be costly. Thus, soil removal must be prevented before it becomes a problem.
- On terraces severely affected by erosion, tillage and cropping systems may need to be shifted. Conservation tillage practices such as no-till and reduced tillage which leave large amounts of residues are recommended practices to reduce erosion within terraces (Fig. 11.18). High-biomass producing-forages and crops also protect soil surface from erosion and reduce the frequency of terrace maintenance.
- Interrill and rill erosion that may occur between terraces must be reduced with the use of conservation tillage. Ridge tillage is used as an alternative to terraces to retain runoff. Ridging on the contour creates mini terraces.
- Terraces must be properly designed to reduce interference with machinery traffic, cropping systems, and field management.

Fig. 11.18 No-till combined with terraces (Courtesy USDA-NRCS)

- All tillage operations must be performed on the contour parallel to terraces to reduce soil erosion. Contours retain runoff, increase water infiltration, and reduce soil loss.
- Terraces under intensive tillage and steep slopes require more frequent maintenance.
- Field equipment and animal traffic over the ridges must be controlled to reduce settlement. Ridges are susceptible to settlement or subsidence so overbuilding of ridges is desirable. Settlement amount depends on the equipment and material used and magnitude of compaction during construction. Ridges built from stones and rocks are less susceptibility to settlement than those consisting of earthen bunds, which settle by more than 10% following construction. Terraces are damaged by rodent activities and natural settling. On broad-base terraces, frequent crossing of ridges with machines causes settling and flatten the ridges.
- Poorly designed channels also pond or perch water, impeding trafficability and crop establishment. If possible, the deposited sediment above ridges must be removed and spread back in the field and used for reinforcing the ridges.
- Grass in the waterways must be mowed and the accumulated sediment removed.
- The underground outlets must be maintained free of any debris and sediment, and established in a depression.
- The riser height should be <2 m and protected with stones, rocks, and compacted soil with permanent vegetation.

11.8 Gully Erosion Control Structures

Restoring gullies is more complicated and expensive than reducing interrill and rill erosion. Once gullies are formed, large amounts of soil from surface and subsurface layers have been already lost. These vertical cuts with depths >5 m prohibit cultural operations and equipment traffic and affect the stability and aesthetics of the whole landscape. No soil erosion type is as visible and dramatic as gully erosion. Thus, preventing erosion is more economical than reclaiming an eroded soil. Intensive grazing, forest fires, deforestation, and intensive cultivation are the main drivers of accelerated gullying.

Preventative measures. Preventative measures of gully erosion include the use of conservation tillage, grass buffers, contouring, strip cropping, and terracing. Maintaining a permanent vegetative cover is the first measure to reduce erosion. Gully formation starts when runoff concentrates in shallow bare depressions and forms isolated rills cutting through the field. Thus, any measure that reduces rill erosion, runoff concentration, and flow channelization prevents gullying. Any restorative strategy must identify first the cause of the gully and then develop counteractive measures. Timing is also an important factor to be considered when establishing erosion control measures. All vegetative and mechanical control measures must be established during dry seasons.

Control options. When gullies are already formed, there are two options of control:

The first option is to reclaim the gullies to conditions similar to the uneroded portions of the field. It involves refilling the channels and reshaping the surface with soil from neighboring fields. The earth removal from adjacent fields may, however, significantly reduce the topsoil depth. Thus, refilling is more appropriate for shallow gullies. Restored gullies must be planted preferable to permanent vegetation, and concentrated runoff causing gully erosion must be controlled.

The second option involves managing the existing gullies by stabilizing the gully head, bed and sides and reducing their expansion. This option is appropriate for large gullies where the cost of refilling and bringing back to their original condition may be higher than the land value following reclamation

Strategies for preventing, restoring, and managing gullies include (FAO, 1986):

- Reduction of runoff peak rates with the establishment of structures that intercept runoff, absorb its erosive energy, and release it at low velocities.
- Diversion of concentrated runoff above the lower points of the fields before it develops new or expands the existing gullies.
- Stabilization of gully bed and sides if the previous two strategies are insufficient to control erosion. In regions with severe gullying, a combination of various measures must be used.

Factors influencing the design of structures. Designing a structure for gully erosion control involves determining the:

- Runoff peak rate
- Flow rate in the gully
- Gully size (width and length)
- Gully networks
- Soil slope
- Soil hydrology
- Drainage area
- Land use and management

The design runoff peak rate is estimated using the rational formula (Eq. 2.33) based on the rainfall intensity, soil condition, and drainage area, while the runoff flow rate (Q) in the gully is estimated using

$$Q = AV \tag{10.19}$$

where V is velocity of the flowing water (m s^{-1}) at any point within the gully and A is cross-sectional area of the wetted portion of the gully (m). The Manning's formula is used to estimate the V of runoff in the gully (FAO, 1986) as

$$V = \frac{1.486 R^{\frac{2}{3}} S^{\frac{1}{2}}}{n} \tag{10.20}$$

where R is hydraulic radius of the wetted area of the gully (m), S is slope gradient of the gully channel (m m^{-1}), and n is roughness coefficient of the channel. The values of *n* vary depending on the type of channel such as earth or lined channels. Channels lined with earth, rubble or stones have an *n* value of about 0.025, while channels lined with concrete, smooth metal, and wood flumes have *n* values of about 0.012.

11.8.1 Types of Structures

Gully erosion is controlled using *temporary* and *permanent* structures, depending on the severity of erosion (Figs. 11.19–11.22). *Temporary structures* include small dams of tree branches, logs, stones, and earthen dams, which are constructed across gullies in series at short intervals along the channel. The availability of construction material (wood, stones) determines the type of dams to be constructed. Use of surface mats is another practice to shield freshly reshaped gullies until permanent vegetation is grown. *Permanent structures* include stone and concrete channel revetments, farm ponds, concrete dams, gabions, grass waterways, drop structures, chutes, and pipe spillways. Lining of gullies is a recommended measure in soils with slope gradients $<25\%$, while installation of permanent drop pipe inlets and other sophisticated structures are necessary in severely eroded soils with slope gradients $>25\%$ to safely convey large volumes of runoff.

Various combinations of control practices are used to stabilize a network of gullies (Fig. 11.23).

Above gullies, diversion systems are established to redirect runoff to stable vegetated waterways. Straw bales placed across channels stabilize the soil (Fig. 11.24). At the lowest section of the main gullies, establishment of more sophisticated

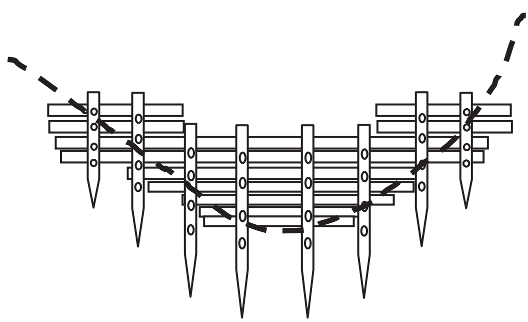

Fig. 11.19 Log dam installed across a gully

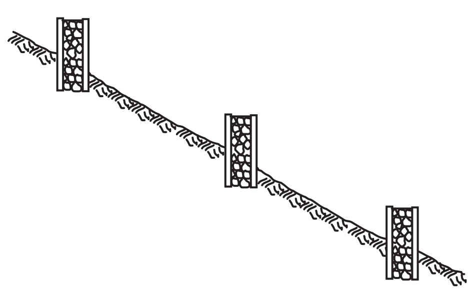

Fig. 11.20 Cross-section view of a log dam filled with stones

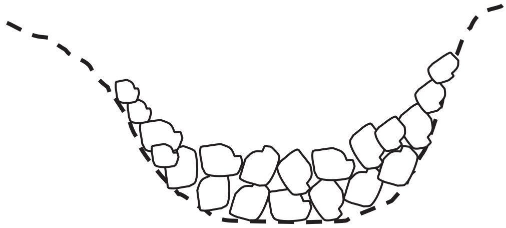

Fig. 11.21 Check dams with stones

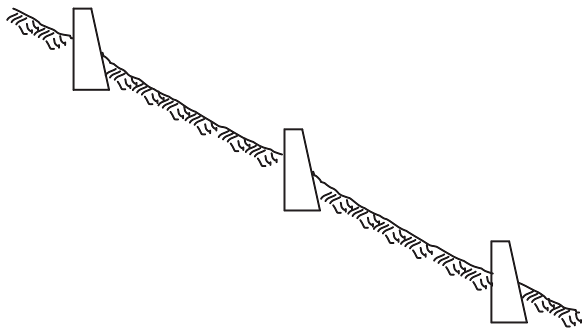

Fig. 11.22 Cross-section view of concrete dams installed in series along a gully

structures may be necessary to control gullying. Two common types of drop structures used at the base of gullies are: *chute spillway* and *graded structures*. Structures that are established across gullies are known as grade structures. Large gullies must be stabilized with combined measures including stone and check damns after they are graded in their sides. These structures reduce runoff velocity by absorbing runoff energy in vertical drops along the channel. Concrete drop structures and gabions are examples of grade structures.

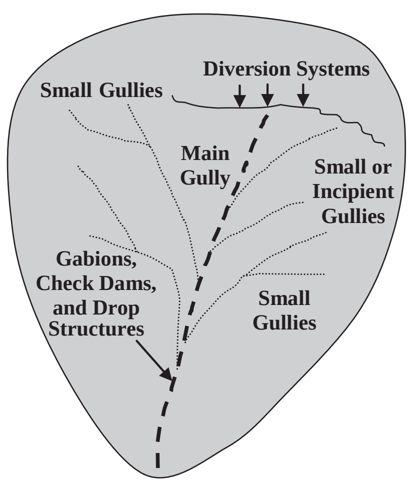

Fig. 11.23 A gully system showing a network of gullies (After FAO, 1986)

Fig. 11.24 Straw bales are used to stabilize waterways (Courtesy Ryan Bartels)

11.8.2 Grassed Waterways

Grassed waterways are wide and shallow grassed channels under perennial grass established along the natural drainage pathways to convey runoff at low velocities and are appropriate for slopes up to 5%. Grassed waterways are often combined with drop structures established at various points within the waterways for reducing the slope. Vegetated waterways are the first choice for controlling gully erosion because they are part of biological practices which mimic the natural field conditions. The width and length of grassed waterways depend on the drainage area and runoff volume. A typical waterway is about 3 m wide and 0.5 m to 1.5 m deep with side slopes of 10%. Geotextile mats and fast growing grass species are used to ensure rapid protection of the channel in newly established grassed waterways. Grass waterways are also used to carry runoff from diversion systems, terraces, field outlets, and culverts along roads. Further discussion on the design of grassed waterways is presented in Chapter 9.

11.8.3 Gabions

Gabions are permanent structures consisting of rocks and stones wrapped in metallic fences and stacked atop one another and are part of the traditional techniques used for retaining walls, protecting culvert headwalls, stabilizing dams, dikes, and channels. These structures reduce gully erosion by reducing runoff velocity, by promoting sedimentation, and by reducing flow channelization. The advantage of using gabions over concrete structures is their relative flexibility and natural adjustment to moderate changes in soil or foundation settlement. Concrete or rigid structures resist compressive forces but fail under high tensile loads unlike gabions.

The gabions are built in three forms: basket, mattress, and sacks, but all consist of cobble- and small boulder-filled baskets trapped with hexagonal mesh of galvanized steel wire. The gabion baskets are rectangular structures with dimensions of about 1 m height and 1 m width with a length of 3–5 m that are staked on top of one another like bricks in a nearly vertical structure along streambanks and at the bottom of sloping fields. The mattress is normally shallow (0.2–0.5 m) and is used to stabilize channel beds. The sacks are simple mesh sacks filled with stones. The length and height of gabions vary depending on the use. Gabions are primarily used for stabilizing steep slopes with non-cohesive or sandy materials in spots where other control practices of less complicated nature (e.g., vegetative cover, surface mats, silt fences) have failed to control erosion.

The structure is built in such a way that the flattest portion of the rock/stones faces the front and is in contact with the wire mesh to ensure long-term durability and improve landscape aesthetics. The rocks interlocking each other and stones filling the interspace voids provide internal stability and firmness to the structure. Gabions are used as outlet structures placed in gully-affected areas, valleys or swales, and steep vegetated channels to dissipate the erosive energy of runoff and reduce concentrated runoff erosion and seepage. Gabions are permeable so that the upland runoff flows through and above the structure, yet are stable structures for providing robust protection against gully erosion, landslides, and mudflow. The heavy wire mesh prevents the basket of rocks from bulging. Gabions are, however, relatively expensive measures and require heavy equipment for transporting large volume of stones. Manual filling of the basket is preferred over mechanical filling to ensure proper and uniform filling material of each rock or stone. Gabions can also be underlined with geotextile fabric to reduce runoff from undercutting below.

The permissible shear stress and thickness of the gabions are estimated as follows (Kilgore and Cotton, 2005):

$$\tau_p = F^* (\gamma_s - y_w) D_{50} \tag{10.21}$$

$$\tau_p = 0.009 (\gamma_s - y_w)(MT + MT_c) \tag{10.22}$$

where τ_p is permissible shear stress (N m^{-2}), F* is the Shields' parameter (dimensionless), γ_s is specific weight of the stone (N m^{-}), γ is specific weight of water (N m^{-3}), D_{50} is median diameter of the stone between 0.076 and 0.457 m (m), MT is gabion mattress thickness (m) ranging from 0.152 to 0.457 m, and MT_c is thickness constant (m) equal to 1.24. The average rock or stone size (d_m) required to build a gabion mattress is estimated (Freeman and Fischenich, 2000) as

$$d_m = S_f C_s C_v d \left[\left(\frac{\gamma_w}{\gamma_s - \gamma_w} \right)^{0.5} \frac{V}{\sqrt{gdK_1}} \right]^{2.5} \tag{10.23}$$

$$C_v = 1.283 - 0.2 \log \left(\frac{R}{W} \right) \tag{10.24}$$

where C_s is stability coefficient (0.1), C_v is velocity distribution coefficient, d is flow depth, g is acceleration due to gravity, K_1 is side slope correction factor, $R$is centerline bend radius of the main channel flow, S_f is safety factor, V is flow velocity, and W is water-surface width.

11.8.4 Chute Spillways

Chute spillways are specifically designed to control overfalls within gullies and grassed waterways (Fig. 11.25). These structures are constructed using concrete blocks, gabions mattresses, rock ripraps, geotextile revetments, and wooden materials to transport concentrated runoff water down steep slopes and convey it at reduced velocities. The chute spillways capture and absorb the energy of concentrated runoff through its sills and wing walls. Chute spillways made of concrete are not as flexible as gabions and their performance may be diminished by seepage and foundation settlements. Chutes are also used to carry runoff water from fields to ditches at low velocities, and their capacity is controlled by the upslope inlet size. These structures are appropriate for slopes up to 25%. Because chutes lined with concrete material may accelerate runoff in steep slopes, rock ripraps or gabions with high roughness are alternatives to reduce erosion downstream. Deceleration structures of rocks ripraps or stone dams improve performance of chutes.

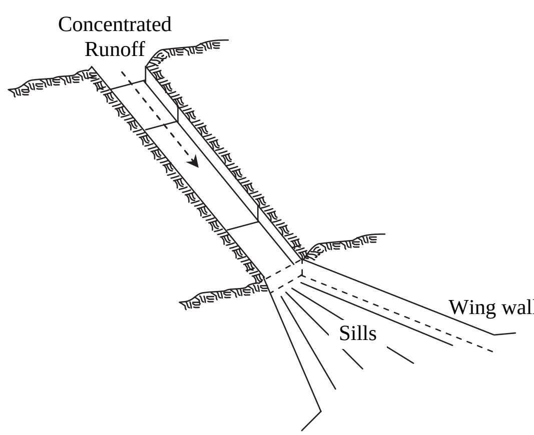

Fig. 11.25 Chute spillway established along a sloping field

11.8.5 Pipe Spillways

These structures are designed for high drops of water runoff. Pipes consist of corrugated plastic and metal pipes with various forms of inlet and outlets (Fig. 11.26). A temporary runoff storage area is required before releasing runoff through the pipes. The area around inlet and outlets must be lined with concrete or stones and/or

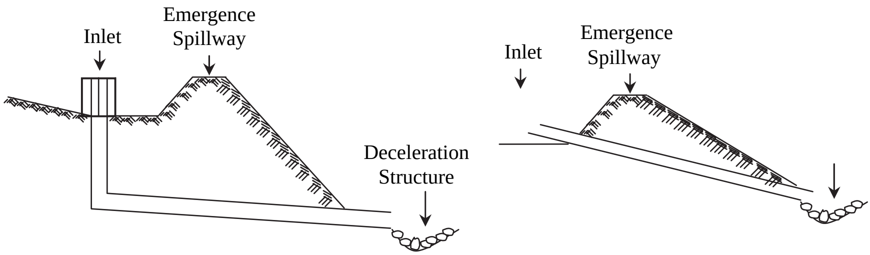

Fig. 11.26 Two types of pipe spillways

compacted material and maintained clean. A pipe spillway is designed from earthen dam constructed across gullies with a corrugated pipe. Water released from pipes is used in ponds and sediment basins. Construction of a spillway is less costly than chutes and drop structures.

11.8.6 Drop Structure

Another gully erosion control practice is the installation of a drop structure (Fig. 11.27). This permanent structure absorbs concentrated runoff and reduces gully erosion by directing runoff through a well-designed drop spillway. Runoff flowing over the drop structure is released into a nearly flat apron before it is carried to a stable channel. The apron absorbs and reduces runoff velocity. The rate of runoff passing over the structure is controlled by a box- or straight-inlet spillway with a depth between 0.5 and 1.5 m. The drop structure is appropriate for low fall heights and occupies less space than other structures and is constructed with concrete, rocks, lumber, or gabions. The drop structures are established at the gully head or at the lower end of gullies stabilized with grassed waterways. Well-designed and stable drop structures carry large volumes of runoff and their performance is not affected by clogging up with sediment and debris. Compared to chutes and pipe spillways, drop structure may be relatively easier to build.

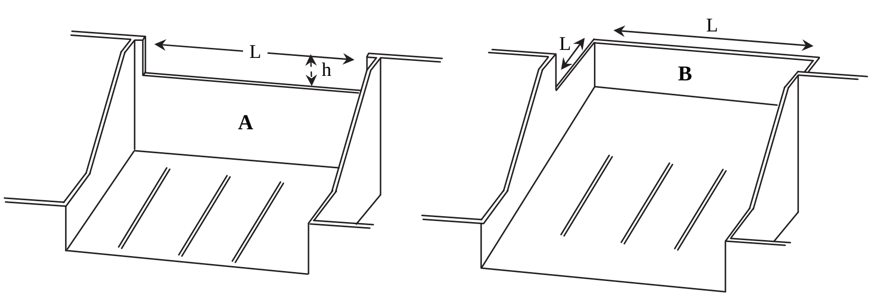

Fig. 11.27 Concrete drop structures with straight- (**A**) and box-inlet (**B**) spillways (Not to scale)

Flow through the drop spillway is computed by the weir equation:

$$q = 0.55CLh^{3/2} \tag{10.25}$$

where q is maximum discharge ($m^3\ s^{-1}$), C is spillway coefficient, L is spillway length (m), and h is depth of spillway (m). A C value of 3 is used for rocks and logs and 1.8 for gabions and concrete structures (Schwab et al., 1993).

Example 5. Compute the maximum flow capacity of a straight-drop spillway of a concrete drop structure constructed as a check dam. The crest length is 3.5 m and flow depth is 0.8 m.

$$q = 0.55CLh^{3/2} = 0.55 \times 1.8 \times 3.5 \times (0.8)^{3/2} = 2.48\ \mathrm{m^3\ s^{-1}}$$

For a box-inlet drop spillway, the crest length is equal to the sum of the three sides of the box inlet.

Example 6. Design a box-inlet drop spillway for a check dam to control gully erosion from a watershed which has an area (A) of 20 ha. The dam will be constructed from gabion baskets. The average 20-min rainfall intensity (i) for a return period of 20 yr for the region is 75 mm h^{-1} where an average of 70% becomes runoff.

Using Eq. (2.33), first compute the peak runoff rate (q) in $m^3\ s^{-1}$.

$$\mathrm{A} = 12\,\mathrm{ha} = 0.2\,\mathrm{km}^2$$

$$q = \frac{C \times i \times A}{3.6} = \frac{0.7 \times 75 \times 0.2}{3.6} = 2.92\,\mathrm{m^3\ s^{-1}}$$

If the C coefficient for concrete structures is 1.8, the dimensions of spillway are calculated as follows

$$2.92 = 0.55CLh^{3/2} = 0.55 \times 1.8 \times L \times h^{3/2}$$

The depth of flow should be between 0.5 and 1.5 m, so assume 0.75 m.

$$2.92 = 0.55 \times 1.8 \times L \times (0.75)^{3/2}$$

$$2.92 = 0.643 \times L$$

$$L = 4.50\,\mathrm{m}$$

The box-inlet drop spillway should have a total length of 4.5 m with two sides of 1 m and the one side perpendicular to slope of 1.5 m.

11.8.7 Culverts

Culverts refer to any circular, elliptical, or box cross section structure under the roadways with an opening large enough (about 3–5 m) to carry high volumes of runoff. These are permanent structures of concrete boxes or corrugated metal pipes. Culverts must be designed to convey peak runoff rates and concentrated runoff from urban areas or farmlands. The amount of runoff carried by a culvert when completely full of water is estimated using (Schwab et al., 1993):

$$q = \frac{a\sqrt{2gH}}{\sqrt{1 + k_e + K_b + K_c L}} \tag{10.26}$$

where q is flow capacity (m^3 s^{-1}), a is conduit cross sectional area (m^2), H is hydraulic head (m), K_e is loss coefficient at the inlet, and K_b is loss coefficient due to roughness (bends) of the culvert. If the culvert is partially filled with water, the flow capacity is computed as

$$q = aC\sqrt{2gh} \tag{10.27}$$

where a is the cross-sectional area (m^2), h is the head to the center of the orifice (m), and C is a coefficient of the conduit equal to 0.6.

11.8.8 Maintenance of Gully Erosion Control Practices

Gully erosion control practices like any other erosion control practice must be maintained regularly. Grassed waterways, chutes and pipe spillways need regular checks for sediment build-up. Settling of the foundations and overtopping with runoff of the permanent concrete structures are causes of failure. The inlet of structures should be maintained free of debris, snow, and ice to accommodate peak runoff rates during spring. The earthen dams must be designed based on the runoff peak rates to reduce overtopping. Soil berms are relatively inexpensive to build but are the most susceptible to failure, causing inundation and flooding of downstream fields. Settlement, shifting, rodent activities, and cultivation of berm backslopes cause failure. Backslopes of dams must be planted to perennial grass species and mowed regularly to maintain a good stand. Sedimentation and runoff cutting alongside the culverts, runoff bypassing drop structures, and seepage and lateral flow around structures may undermine the performance of structures. Structures must be keyed to the lateral banks to resist undercutting and lateral flow. Clogging of pipes in spillways with debris and sediment must be controlled. Sophisticated designs of permanent structures are discussed in detail by Schwab et al. (1993).

Summary

Mechanical or engineering structures are an important component of soil erosion control measures. These structures are used in conjunction with biological control practices (e.g., residue mulching, no-till, reduced tillage, cover crops, riparian buffers, grass filter strips) to control soil erosion. While biological measures are more economically feasible and more environmentally friendly than engineering structures, soils under severe erosion require the establishment of structures to intercept large volumes of runoff and sediment. Mechanical structures reduce runoff velocity, pond runoff water, convey runoff at non-erosive velocities, trap sediment and nutrients, prevent flooding of neighboring lands, and reduce sedimentation of downstream water sources. Engineering structures may be permanent (e.g., terraces, drop structures, spillways, culverts, gabions) on temporary (e.g, contour bunds, sand bags, silt fences, surface mats, log barriers). Their choice depends on the severity of erosion, soil slope, and climate.

Farm ponds are also structures that are constructed to control erosion and flood. Ponds are normally established in depressions or at lower points within watersheds to capture upstream runoff. These structures collect runoff, store water, and have a multi-purpose use. Ponds provide water to livestock, irrigation, and wildlife habitat while reducing runoff formation and soil erosion. There are various types of ponds including rain-fed, groundwater-fed, stream-fed, and off-stream ponds.

Terraces are earthen embankments established across the dominant slopes and are common to sloping terrains. They modify the land topography and collect and divert runoff to outlets without causing erosion. When used in conjunction with conservation practices, terraces are one of the most effective structures to control soil erosion. Terraces can be: broad-base, narrow-base, bench, and steep backslope terraces. Structures for controlling gullies are more expensive than those for reducing interrill and rill erosion. Stone and concrete channel revetments, concrete dams, gabions, grass waterways, drop structures, chutes, and pipe spillways are used for gully erosion control.

Study Questions

1. Determine the cut and fill volume of soil for a 150 m terrace with balanced cross section on a 5% slope if the channel depth is 0.40 m. The freeboard is 0.06 m.
2. The soil loss for a sloping in Ohio is 13 Mg ha^{-1}. Determine the slope length and terrace spacing needed to reduce loss of soil from this field by 50% if the USLE factor values are K = 0.15, L = 140 m, S = 7%, C = 0.25, and P = 0.4.
3. Determine the terrace spacing for a highly erodible soil in Missouri with a 5% slope (Y = 0.3). Determine the even multiple of turns between terraces if the width of the field equipment is 6 m.
4. Compute the flow capacity of a box-inlet drop spillway that has a crest of 1 m in two sides and 2 m in one side for a depth of flow of 1.2 m.

5. Determine the crest length of a box-inlet drop spillway to be installed along a gully to transport runoff from a 30 ha watershed for a rainfall intensity of 80 mm h^{-1} if 80% of rainfall becomes runoff.
6. Describe the different types of terraces.
7. Discuss how ponds should be designed and maintained.
8. Describe the structures to control gully erosion.
9. Discuss differences between biological methods and mechanical structures for reducing non-point source pollution.
10. Describe the types of terraces used in mountainous regions.

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Chapter 12
Soil Erosion Under Forests

A forestland is a non-agricultural land with at least 10% of tree cover (FAO, 2000). In this context, the global forest area is estimated at 4 billion ha or about 30% of the total terrestrial surface (FAO, 2005). Forests are not evenly distributed across all regions and are rather concentrated in specific climatic zones. Depending on the climate, forests are classified as *temperate, boreal*, and *tropical* forests. Depending on the growth characteristics, forests are classified as *evergreen, semi-evergreen, deciduous, lower and upper montane, mixed, and mangrove* forests. Extensive forestlands in the world include rain forests in the Amazon, Congo Basin and Sumatra, dry woodlands in Southern Africa, coastal mangroves in Southeast Asia; and alpine forests in the Andean region of South America.

12.1 Importance of Forestlands

Natural forestlands are important to conserve soil and water, sequester C, and mitigate net emissions of greenhouse gases while providing wood, fuel, food, fodder, medicines, and other products (e.g., dyes, tannins, perfumes, ornamentals, exudates) (FAO, 2000). The capacity of a forest to produce tree-based products is termed forest productivity. Forests moderate climate by affecting fluctuations in temperature, relative humidity, evaporation, radiation, and desiccation. Thus, forests influence global climate, conserve biodiversity, and improve environmental quality. A natural forest is the most biologically diverse ecosystem and is thus a natural reserve of genetic diversity of flora and fauna. Rainforests, for instance, are habitat for millions of plant and animal species.

Forests also play a major role in moderating soil and ecosystem hydrology and water balance. Trees capture, absorb, store, and distribute water in the soil. Water dynamics such as precipitation storage and distribution, evaporation rates, overland flow, drainage, saturated and unsaturated lateral flow, interflow, baseflow, and leaching are all influenced by forest cover and its management. Availability of freshwater is decreasing worldwide due to rapid population growth. Scarcity of freshwater is the most severe in tropical and arid regions. In this context, rainforests represent a permanent reservoir of freshwater and play an important role in quantity and quality of renewable freshwater resources in a watershed.

12.2 Classification of Forests

Forests can be classified into four main categories: *primary, modified, semi-natural, and forest farms* (Fig. 12.1; FAO, 2005).

1. ***Primary forests*** are natural forests predominated by native species with little or no human disturbance or intervention. A natural forest can be *open* or *closed* (Lal et al., 2004). *Closed* forestlands are dominated by trees and are ungrazed, while *open* forestlands consist of both trees and grasses, and are often grazed.
2. ***Modified natural forests*** represent the largest area of forestlands and are managed with selective logging and regeneration of native species.
3. ***Semi-natural forests*** consist of intensive harvesting followed by seeding and assisted natural regeneration.
4. ***Forest farms*** include planted trees for wood, fuel, fiber, and bioenergy production. While most forest farms are aimed at producing commercial forest products, there is an increasing trend to grow trees for soil and water conservation (e.g., agroforestry). Short-rotation woody species are prime source of feedstocks for biofuel and paper industries.

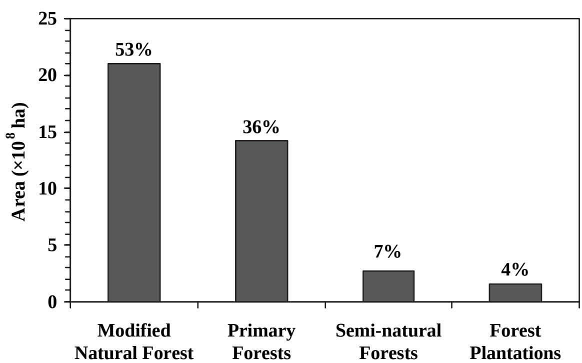

Fig. 12.1 Classification of world's forestlands (After FAO, 2005). Percent values indicate the relative proportion with respect to total forest area

12.3 Natural Forests and Soil Erosion

Undisturbed perennial forestlands generally produce the least amount of runoff and soil erosion among all land use systems. Soil erosion from undisturbed forest soils normally ranges from 0.02 to 1.2 $\mathrm{Mg\,ha^{-1}}$ (Wagenbrenner et al., 2006). Natural forests reduce soil erosion by forming a dense and multistory canopy with thick forest floor litter and extensive root system. These characteristics capture and sponge up raindrops, store rainwater, and release water through seepage at non-erosive velocities. The dominant canopy cover reduces the rainfall erosivity and protects the soil from the direct impact of raindrops and throughfall. Raindrops are intercepted by the leaves and branches (canopy interception) and flow down the trunks (stem flow) at reduced velocity.

12.3.1 Canopy Structure

Structural arrangement of forest vegetation influences water balance and rates of erosion. Single-storey forest with limited undergrowth from high temperatures and reduced light may not reduce runoff and soil erosion as much as multistorey forests. Raindrops intercepted by (>8 m) tall trees in single-storey forests often regain their terminal velocity, causing severe soil erosion. Moreover, concentrated streams of raindrop water flowing along tree trunks can lead to land slides and slope wash. Multistorey canopy cover reduces the terminal velocity of raindrops and the attendant soil erosion. Forests or woodlots are also important in reducing wind erosion (e.g., windbreaks). Soil erosion rate and amount are influenced by percent canopy cover, deforestation, and climate (e.g., rainfall distribution).

12.3.2 Forest Litter and Roots

Forest floor refers to continuous litter and unincorporated humus remaining on the soil surface. The combined effect of tall vegetation and dense litter cover is to buffer raindrop impacts and reduce soil splash and detachment. Deep tree roots enhance water infiltration rate, improve soil macroporosity, increase soil organic matter content, increase drainage, and reduce runoff and soil erosion rates. The dense forest litter slows runoff velocity and filters sediment and chemicals. Root channels constitute pathways for rapid downward water flow, and their disturbance increases runoff and soil erosion rates. Forest litter is a natural mulching material that conserves water, reduces evaporation, moderates soil temperature, improves soil structure, and promotes tree growth.

12.4 Deforestation and Soil Degradation

Clearing of forestlands, known as deforestation, eliminates the innumerable environmental benefits of forests (Fig. 12.2). Deforestation is responsible for the rapid decline in the extent of forests particularly in tropical regions. Conversion of indigenous forest vegetation to agricultural systems (e.g., croplands and pasturelands) is an example of accelerated deforestation. Deforestation impoverishes rural dwellers particularly in tropical countries with an economy strongly dependent on forest resources. Deforestation does not always imply a complete clearing of trees. A deforested land may still contain trees but the forest attributes (e.g., number of species, growth, production potential) are altered. A deforested land is a system whose structure and function have been altered by natural and/or human-induced

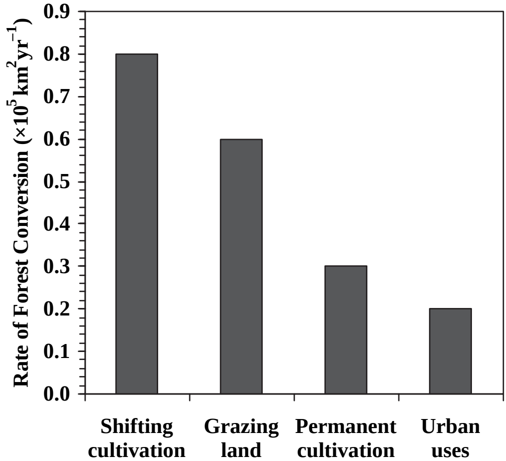

Fig. 12.2 Estimated rate of annual conversion of tropical forestlands to other land uses (After FAO, 2005). Primary forests are being converted to modified or semi-natural forests in tropical regions. Extensive evergreen forest landscapes have been impoverished and turned into eroded lands in mountainous forestlands

causes. Forestlands, similar to pasturelands and rangelands, are influenced by numerous factors which reduce their ability to provide all the ecosystem services including diversity of flora and fauna and protection of the environmental. It is estimated that about 13 Mha of forest are cleared annually worldwide. South America (4.3 Mha) and Africa (4.0 Mha) experienced the greatest loss of forest between 2000 and 2005 (FAO, 2005).

12.4.1 Soil Erosion

The leading cause of degradation of deforested soils is erosion by water and wind. Human-induced disturbances such as tree removal, grazing of woodlands, fires, and road constructions set the stage for accelerated soil erosion and water pollution with sediment and nutrients. Amounts of runoff and soil erosion are a function of the rate and magnitude of forest disturbance. Because runoff and soil erosion are functions of vegetative cover, removal of trees dramatically increases runoff and soil erosion. Soil erosion follows the order: intensive tillage > conservation tillage > pastureland > agroforestry > native forest (Fig. 12.3).

Impacts of deforestation on soil erosion are generally high in steep mountainous terrain used for extractive farming. In mountainous regions of Jamaica, soil erosion from slash-and-burn agriculture was 20-fold higher than that from native forest. In deforested sites in northern Brazil, erosion rates were 115 Mg ha^{-1} from freshly deforested soils, 8.6 Mg ha^{-1} from soils under grass, and only 1.2 Mg ha^{-1} from soils under shrubs and trees (Ramos and Marinho, 1980). Deforestation drastically increases the soil-erodibility factor close to 1 because trees anchor the soil against erosional processes (Celik, 2005). Flooding, siltation, water pollution, and decline in aquatic life (e.g., coral reefs) or coastal fisheries are direct results of deforestation.

High risks of soil erosion in tropical than in temperate regions are attributed to the fact that tropical soils: (1) receive low levels of radiation, (2) have relatively

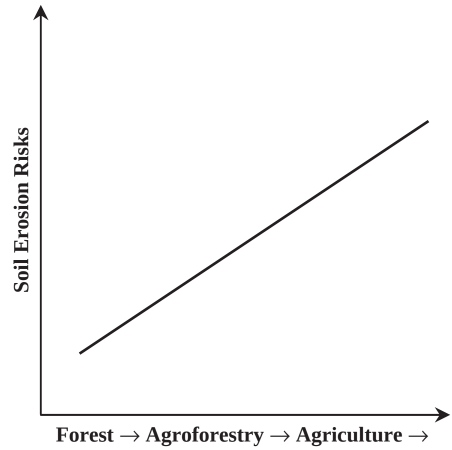

Fig. 12.3 Runoff and soil erosion increase with land conversion to intensive agriculture as the soil protective cover as well as the below-ground biomass (e.g., tree roots) which anchor and stabilize the soil are removed

thin forest floor, (3) contain low soil organic matter content, (4) receive intense and highly erosive rainstorms, and (5) are under rugged terrains with steep slopes (Lal, 1996). Most of the runoff and eroded material from forestlands occur in deforested or disturbed areas (e.g., gravel roads, paved roads, log landings, fire-affected areas, trails). Surface cover determines erodibility of forested soils (Fig. 12.4). The same forestlands under different levels of tree harvesting or fire have different soil erodibility because of differences in the amount of surface cover. For example, soil hydraulic conductivity under forest is commonly much higher than under croplands, but that under forest roads is often lower than under croplands (Elliot, 2004).

Forest roads, tracks, and skid trails are potential sources of runoff and soil erosion in addition to taking significant portions of forest land out of production. Roads and network roadside channels concentrate runoff, develop gullies, accelerate storm discharge, and speed up delivery of sediment and nutrients to downstream water sources. Loose soil material along roads is readily eroded and transported in runoff. Bare and less cohesive soils are the most susceptible to rilling and soil loss. Soil erosion from roads and skid trails constructed on steep forest hillsides increases with increase in slope gradient and width of pathways. Soil erosion from forest roads decreases, however, with time after construction. On a tropical forest in Malaysia, soil erosion rates decreased from 13.3 Mg ha^{-1} to 3.1 Mg ha^{-1} for roads and from 10.1 Mg ha^{-1} to 2.1 Mg ha^{-1} for skid trails from the first to second year following construction, respectively (Baharuddin et al., 1995).

12.4.2 Soil Properties

Conversion of forests to crops and pastures generally degrades soil properties important to agronomic productivity (Table 12.1). Excessive cultivation following deforestation may reduce quantity and quality of soil organic matter, accelerate its decomposition, and reduce biological activity and diversity with the attendant rapid

Fig. 12.4 Deforestation creates bare areas with soils highly susceptibility to erosion (Courtesy Rhett A. Butler)

deterioration in nutrient cycling and storage. Mechanized land clearing through soil disturbance disrupts aggregate formation, increases wettability, reduces macroporosity, reduces the proportion of macroaggregates, and increases soil's susceptibility to erosion. Stable macroaggregates are essential to withstand erosive forces of water and wind. Deforested soils are prone to compaction and have low plant available water content, water infiltration rates, and saturated and unsaturated hydraulic conductivities. Structurally stable soil is rendered unstable and prone to erosion. Deterioration in surface soil properties results in increased risks of runoff and soil erosion, leading to pollution and euthrophication of downstream water resources.

Use of heavy machines for clearing and post-clearing cultivation scrapes surface soil and exposes subsoil with unfashionable soil properties. Accelerated erosion may lead to increase in sand content and reduction in clay and silt contents in deforested lands. Runoff from structurally degraded soils preferentially carries clay and fine silt particles, leaving sand particles behind, and resulting in textural modification.

Table 12.1 Deforestation effects on selected soil properties in the top 10-cm depth

Soil properties	Deforestation	
	Before	After
[1]Bulk density (Mg m^{-3})	0.9a	1.4b
[1]Cone index (MPa)	0.2a	0.7b
[2]Aggregate mean weight diameter (mm)	7.4a	4.9b
[3]Saturated hydraulic conductivity (mm h^{-1})	$>$166a	46b
[3]Water infiltration rate (cm h^{-1})	334a	67b

[1]Blanco-Canqui et al. (2005), [2]Hajabbasi et al. (1997), and [3]Lal (1996).

Clay eluviation with percolating water is also a contributing factor to development of a coarse-textured soil surface.

12.5 Causes of Deforestation

As shown in the list below, anthropogenic activities are the principal causes of deforestation.

1. Clear-cutting and agriculture
2. Grazing
3. Tree harvesting
4. Wildfires
5. Mining
6. Development of petroleum industries
7. Urbanization and new settlements

12.5.1 Cultivation

Agricultural expansion is the main driver of deforestation. It fragments forestlands and degrades converted lands. In tropical countries, shifting cultivation through slash-burn agriculture is a common practice to produce crops. Extractive farming causes rapid degradation of soil and thus new areas are brought into cultivation, which unravels the cycle of deforestation and soil degradation. Complex and multi-species forests have converted into monocropping systems. Rapid increase in large and commercial farms has accelerated deforestation in developing countries. Deforestation in temperate forests is not as drastic as that in tropical forests since the relatively fertile soil in temperate regions favors rapid vegetation growth.

12.5.2 Grazing

Conversion to pastures is another factor responsible for deforestation. Large tropical forests have been converted to pasturelands in Central and South America. In Brazil alone, nearly 72% of forest clearing was due to livestock grazing or ranching. High demands for livestock production are major factors leading to deforestation. Animals also disturb the forest soils by trampling and creating pathways of access. It reduces the amount of forest litter and increases the percent of bare soil by removing the aboveground biomass. Selective grazing reduces biological diversity and alters the forest composition and structure. Large and localized concentration livestock production systems exert the greatest pressure on forestlands.

Reduction of litter cover reduces the protective effect of forest cover against the raindrop impacts and shearing force of runoff. Soil erosion rates from intensively grazed or degraded forestlands are greater than those from natural forests. Increased runoff and soil erosion reduce growth of under- and over-story vegetation. Grazing and trampling on log landings cause soil erosion and reduce tree regrowth. Excessive grazing deteriorates water quality with pollutants by occasional wading, depositing dung along streams and rivers, and causing trampling and streambank erosion. On sloping forest lands, litter is gradually translocated downhill, which exposes mineral soil and creates bare areas prone to erosion. Excessive grazing and trampling also compact the soil, alter structural stability, and reduce macroporosity, water infiltration and storage, and soil organic matter and nutrient accumulation. The cumulative effects of grazing produce visible damage to soil, increasing soil erosion hazards.

12.5.3 Logging

Demand for wood is increasing at a rate of about 2.6% per year, and tropical forest area is decreasing at a rate of 0.8% per year (FAO, 2000). Heavy logging and road building not only remove trees but also leave behind a system prone to severe soil erosion (Fig. 12.5). Logging operations exacerbate runoff and soil erosion by disturbing soil through wheel traffic, construction of roads, skid trails, dragging of trees, and log landing sites. Complete removal of trees including slash and surface litter causes greater soil erosion than forests with stems removed only.

Two common methods of tree harvesting are: selective and clear-cutting.

- ***Selective tree harvesting*** consists of removing individual trees or groups of trees of the same species for wood and fiber. This practice thins vegetative cover,

Fig. 12.5 Excessive tree cutting for wood fuel and lumber causes soil erosion (Courtesy Rhett A. Butler)

reduces tree density, and disturbs soil during traffic, but its negative impacts are not as dramatic as those from clearcutting. Selective harvesting can be beneficial to reduce competition among tree species for water, light, and nutrients. It is also used for selecting species for seed production.

- ***Clear-cutting*** removes all trees, and branches and leaves (known as “slash”) are often burned. The cleared lands are used for crop and pasture production. This form of harvesting causes the greatest soil disturbance and soil erosion.

12.5.4 Urbanization

The rate of conversion of tropical forestland to urban uses is estimated at about 2 Mha per year (FAO, 2000). About 50% of the total converted area from forest to urban lands occurs in tropical forests. Accelerated demographic changes by industrialization and expansion of urban areas reduce forest and alter the nature of watercourses and quality of water (Fig. 12.6). Uncontrolled runoff from urban impervious surfaces and disturbed construction sites carries sediment and other point-source pollutants.

12.5.5 Wildfires

Uncontrolled fires or wildfires reduce soil and environmental quality (Fig. 12.7). The magnitude of the problem varies across the world. Percent of total forest area affected by fire is the largest in Asia (Fig. 12.8) (FAO, 2005). The frequency of uncontrolled fires has increased with increase in population, and previously burned

Fig. 12.6 Wood cutting and urban development are direct causes of deforestation (Courtesy Rhett A. Butler)

Fig. 12.7 Forest burning causes extensive deforestation (Courtesy Rhett A. Butler and USDA-NRCS)

areas are burned again in many forestlands. For example, the number of forest fires in Italy increased from 3,400 in 1970 to 10,500 in 2006, corresponding to an increase in burned area of 43,000 and 118,000 ha, respectively (Rulli et al., 2006). Wildfires destroy the natural forest structure and change the composition and density of species. Adverse effects are more severe in drylands with limited vegetative cover.

Uncontrolled fires decrease biological diversity, emit greenhouse gases, pollute the environment with smoke, eliminate wood and non-wood products, and burn urban and rural properties. Rehabilitation of burned soils and the resulting treatment of polluted water are expensive. Forest fires damage soil more than pasture/grassland fires since the large forest biomass increases fire residence time and intensity. Forest fires cause dramatic changes in landscape features and soil characteristics.

Soil erosion. Forest fires increase soil erosion by destroying the vegetative cover, which reduces the amount of rainfall interception. Bare/burned soils are exposed to

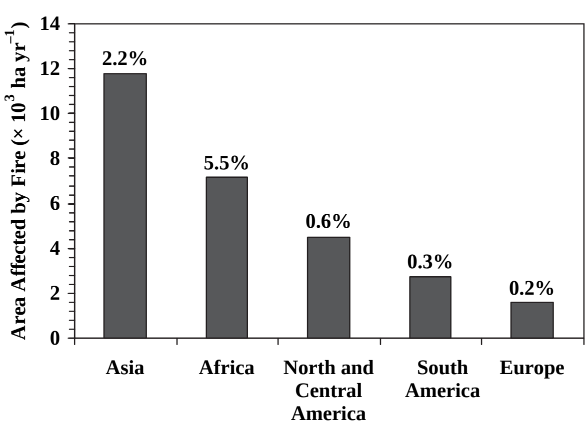

Fig. 12.8 Area and percent of total forest area affected by fire (After FAO, 2005)

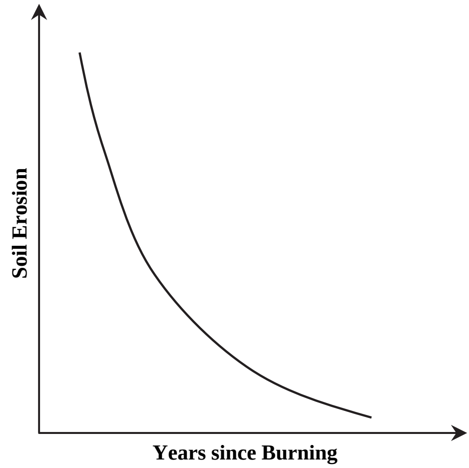

Fig. 12.9 Soil erosion from forest soils decreases with increase in time following fires

the erosive forces of raindrops and overland flow. The extent of burning determines the magnitude at which the soil erodibility is altered. Runoff and soil erosion from burned areas can be 10- to 20-fold higher than unburned forests, increasing with increase in bare soil area and slope gradient. Forest fires increase runoff and soil erosion by several orders of magnitude when followed by seasons of intense rainstorms. Soil erosion is often the highest three or four months after the fire and gradually decreases with soil and vegetation recovery (Fig. 12.9). Soil erosion from burned areas in Colorado was 6 Mg ha^{-1} in the first and second two yr following burning, 1.2 Mg ha^{-1} in the third year, and 0.7 Mg ha^{-1} in the fourth year (Wagenbrenner et al., 2006).

Soil properties. Intensive burning of organic materials degrades soil structure, reduces macroporosity, and causes surface sealing. By burning the forest biomass, fires expose soil surface to direct raindrop impacts, change the surface and subsurface soil hydrology, and alter soil structural properties. Reduced water infiltration and degraded soil structure affect soil chemical and biological properties. Recovery in soil properties is slow and can take more than 10 yr, depending on the severity of fire, resilience of the system, and climate (Moody and Martin, 2001). In Turkey, 8 yr after a wildfire, burned soils had consistently lower aggregate stability, hydraulic conductivity, total porosity, water content, microbial biomass, and soil organic matter content than unburned soils (Ekinci and Kavdir, 2005). Deterioration of soil properties and reduction in nutrient cycling reduce forest productivity.

12.6 Global Implications of Deforestation

Deforestation has ecological, environmental, and agronomic consequences. It alters both cycles of C and fluxes of water, air and heat, thereby influencing local and global climate change. Exposed bare soils are subject to extreme fluctuations in temperature, radiation, and soil water content. Land clearing decreases soil organic matter content and increases losses of ecosystem C. Unless integrated with appropriate soil and water conservation practices, cleared lands for crop cultivation are susceptible to rapid degradation.

A vivid example of the dramatic consequences of deforestation is the Great Red Island, Madagascar, where losses of soil amount to, as much as, 400 Mg ha^{-1} yr^{-1}(Lal et al., 2004) (Fig. 12.10). Land clearing in this region of Madagascar for cultivation has caused the disappearance of about 90% of the natural forest (Lal et al., 2004). Streams and rivers are virtually red and filled with silt, affecting water quality, aquatic life, and biodiversity. Satellital images show that Madagascar's largest lake, Lake Alaotra, has shrunk gradually in the past five decades due to devastating deforestation and its size is now only about 20% of its original size (Bakoariniaina et al., 2006). The reduction in Lake Alaotra's size has affected rice production due to limited irrigation water, high siltation of reservoirs, and continued slash and burn agriculture in neighboring areas of the basin. Soil erosion from deforested hillslopes generally exceeds the threshold levels and leads to denudated bare and steep slopes.

Deforestation interrupts the hydrological cycle, alters water balance, and jeopardizes water quantity and quality. It adversely affects surface and subsurface soil hydrological properties essential to water distribution, groundwater recharge, water retention. It increases risks of flooding and drought. Adverse impacts of deforestation on water dynamics are the strongest immediately after deforestation and decrease with time as a new equilibrium is established through reforestation. Differing amounts of forest litter and quality and quantity of soil organic matter determine water movement and retention.

Fig. 12.10 Erosion is severe in deforested areas in Madagascar (Courtesy Rhett A. Butler)

12.7 Methods of Land Clearing

Methods of land clearing and post-clearing management influence rates of runoff and soil erosion. The methods of removing trees and transporting logs fall into four categories: *low*, *medium*, *high*, and *complete* mechanization levels (Heinrich, 1998). Choice of the level of mechanization depends on the terrain characteristics, accessibility, and costs.

- Low-level mechanization refers to the use of non-industrialized techniques where felling and delimbing of trees are done manually, and logs are transported by rolling (e.g., gravity in steep terrains) or pulling or pushing with animals. These manual operations are being replaced with mechanized techniques of logging and transport.
- The medium level of mechanization combines the use of chainsaws and tractors for harvesting trees.
- The high and complete mechanization levels use sophisticated machines for all harvesting operations (e.g., feeling, extraction, and hauling). Tractors, skyline cables, balloons, and helicopters are some of the modern log transport systems (Heinrich, 1998). Ground-dependent (e.g., tractors) transport systems are, however, less expensive than non-ground dependent systems (e.g., mobile cables, helicopters).

Mechanical removal of trees with large tractors causes more soil disturbance than manual clearing. Methods that disturb the least are also the most expensive techniques of harvesting. Heavy machines compact soil, reduce water infiltration, increase soil erosion, and decrease tree production. Manually cleared forests followed by no-till management produce the lowest rates of soil erosion while those cleared with heavy machines suffer the greatest losses of soil (Lal, 1996). Machine-cleared soils necessitate the use of effective soil and water conservation practices following deforestation including seeding or planting and mulching.

12.8 Water Repellency of Forest Soils

Water repellency refers to the ability of the soil to resist the rapid penetration of water. High concentration of soil organic matter can induce hydrophobic properties to forest soils. Organic exudates, waxes, mucilages, and humic substances form hydrophobic surface films on primary and secondary soil particles (Chenu et al., 2000). These hydrophobic coatings repel or slow water entry into the soil.

Some of the factors affecting water repellency in forest soils include the following (MacDonald and Huffman, 2004):

- Fires
- Forest type (e.g., natural forest)
- Tree species
- Quantity and quality of litter

- Forest growth and age
- Landscape attributes
- Soil texture and clay mineralogy
- Climate (e.g., temperature)
- Antecedent soil water content
- Time since burning

Fires are the predominant factor of inducing excessive water repellency in forest soils. Burning of trees and forest litter releases and vaporizes organic compounds which penetrate into the soil. In contact with the lower soil temperature, these organic compounds condense and coat soil particles, creating water repellent primary and secondary soil particles. Intense fires develop highly water repellent soil layers as more forest litter is burned and soil surface is heated. Intense wildfires burn both forest litter and surface soil to coat water repellent organic substances to soil particle surfaces. Fire intensity, time after forest fire, type of forest, tree species, soil texture, and quantity and quality of soil organic matter influence the natural water repellency of soil.

Soil particles with low specific surface area are more prone to developing hydrophobic surfaces. The more severe the fire, the greater is the water repellency, and lower are the infiltration rates. Loss of vegetation cover and gains in soil hydrophobic properties explain most of runoff discharges ($r^2 > 0.70$) from burned sites (Benavides-Solorio and MacDonald, 2001). Water-repellent soils may not completely block the water penetration but slow the water entry. The water repellency of soils increases with decrease in water content. Fire-induced water repellency is a widespread phenomenon in forest soils, and affects soil and ecosystem hydrology.

Even in the absence of fire, forest soils tend to be more hydrophobic than cultivated soils. While moderate water repellency has beneficial effects on soil structural stability, excessive water repellency can reduce water infiltration and increase runoff rates. The increased runoff often develops rills and exacerbates erosion of soil on sloping forestlands. Frequent and persistent flooding, sedimentation, and water pollution are associated with severe water repellency in forest soils.

12.9 Management of Burned Forestlands

Restoration of burned lands must focus on maximizing the vegetation recovery. Increase in vegetative cover reduces soil erosion and is the single most effective measure to restore degraded soils. Well-established vegetation improves water infiltration, reduces runoff and soil erosion, and protects on-site and downstream productive lands and surface aquatic resources. Speed of recovery depends upon soil and landscape characteristics, fire intensity, time since burning, and climate. Some of the measures for reclaiming burned forest soils include seeding, mulching, and contouring with logs (Robichaud, 2000).

Seeding. It is the most common practice to reestablish vegetation in burned or degraded forest areas (Fig. 12.11). High seed density and good timing under adequate soil conditions (e.g., water content) favor rapid growth and dense vegetative cover. Performance of seeded vegetation is, however, site specific. In some ecosystems, seeded vegetation may compromise the recovery of native vegetation (Keeley, 2006). Artificial seeding is not as effective as natural vegetation and may lead to greater rates of runoff and soil erosion than naturally vegetated areas. A successful establishment of seeded vegetation depends on the soil type, topography, climate, and tree species. Burned soils on steep slopes have lower rates of seed germination due to frequent erosion events and soil instability. Use of native tree species rather than introduced species is an alternative. Seeding with native species or varieties, which are similar to the pre-fire conditions, simulates natural vegetation.

Mulching. By protecting the soil surface and reducing erosion, mulching improves the regrowth of native and non-native vegetation. Mulching is often a temporary measure to control soil erosion from disturbed sites while permanent vegetative cover is established. Crop residues, forest litter, jute, and paper are used as mulch materials. Mulching reduces proliferation of weeds on exposed soils. In some soils, excessive mulch cover may delay regeneration of native species by inducing competition for nutrients, water, and light by the non-native plants. Crop residue mulch must be free of non-native seeds to enhance native vegetation recovery.

Contour logging. Logs placed on the contour in close contact with the exposed sloping soil surface intercept runoff and deposit sediment upstream. Wagenbrenner et al. (2006) developed empirical relationships to estimate the amount of runoff or sediment that can be stored above the contour-felled logs (S_L, m^3 ha^{-1}) placed on burned forest hillslopes (Eq. 12.1) and the actual change in the amount of runoff (ΔQ, m^3) due to contour felling (Eq. 12.2).

Fig. 12.11 Disturbed area seeded to grass under mulch cover (Courtesy Ryan Bartels)

$$S_L = \frac{1}{A} \sum_{i=1}^{n} \frac{Ld}{2} \left(x - \frac{\pi d}{4} \right) \tag{12.1}$$

$$\Delta Q = S_L A_H 100 A_T (\int_0^D I_T dt - \int_0^D I_H dt) \tag{12.2}$$

where A is measurement area (ha), n is number of logs in the measurement area, L is log effective length (m), d is log diameter (m), x is distance (m) from top of the log to the hillslope, A_H is hillslope area (m^2), A_T is area (m^2) of the trenches above the logs, I_T is infiltration rate (cm h^{-1}) within the trenches, I_H in infiltration rate (cm h^{-1}) in the neighboring hillslopes, D is rainstorm duration (h), and t is time (h). Performance of logs for controlling soil erosion from burned forest soils depends on the design, establishment, and maintenance of logs. Overtopping with runoff sediment in intense storms and poor contact between logs and soil surface are common problems. Contour logging is an effective measure to reduce soil erosion under small and moderate storms.

Among the factors that influence the rate of recovery of burned forestlands are:

- intensity of fire (duration, frequency),
- soil properties (texture, water retention),
- vegetation type (forest age, tree species),
- landscape characteristics (slope, geomorphology),
- time since fire event,
- degree of burning, and
- rainfall events (intensity, frequency).

Prescribed forest fires. There are three common types of fires: *wildfires, prescribed forest fires,* and *agricultural burning* (e.g., slash and burn). The later two types involve managed fires and cover small areas than wildfires. A prescribed fire is an important forest management strategy. While excessive burning induces hydrophobic properties to the soil, increasing runoff rates (Robichaud, 2000), well-managed fires control vegetation density, improve distribution of species, enhance forest productivity, and reduce severe wildfire hazards. Some forest systems depend on prescribed fires to rejuvenate and sustain their production. Untimely and mismanaged prescribed fires may, however, cause serious damage to soil, water, and air quality and wildlife habitat (e.g., nesting). Designing frequency, size, intensity of prescribed fires is essential to achieving the forest management objectives.

Well-managed prescribed fires improve forest production for economic and ecologic benefits by:

- reducing forest wildfire hazards,
- eliminating plant competition,
- eliminating weeds and insects,
- improving soil resilience,
- improving grass quality (palatability),
- improving access to forest areas,

- facilitating soil preparation for seeding,
- improving resilience of flora and fauna,
- improving landscape appearance, and
- increasing nutrient recycling (e.g., P, K).

12.10 Reforestation

This practice refers to the reestablishment of trees in deforested lands. It is the best option to overcome deforestation and offset soil degradation. Tress can be reestablished either by natural regeneration or by seeding and transplanting. It takes about 4–5 yr before trees can produce sufficient litter cover to significantly reduce runoff and soil erosion. Litter cover of at least 5 cm depth is required to protect the soil against erosion. Management of young tree plantations is important to reduce formation of rills and gullies, which are likely to occur during the vegetation establishment stage. The global area under forest plantations has steadily increased since 1970s (Fig. 12.12; FAO, 2000).

While reforestation has somewhat counteracted deforestation, its expansion has been slow to effectively off-set deforestation. Reforestation is practiced in small deforested areas. In some places, single tree species for timber production have replaced areas previously under multi-species natural forests. Plantations of single species are used for commercial purposes and not specifically designed to restore biodiversity lost with deforestation. Natural forests provide not only wood,

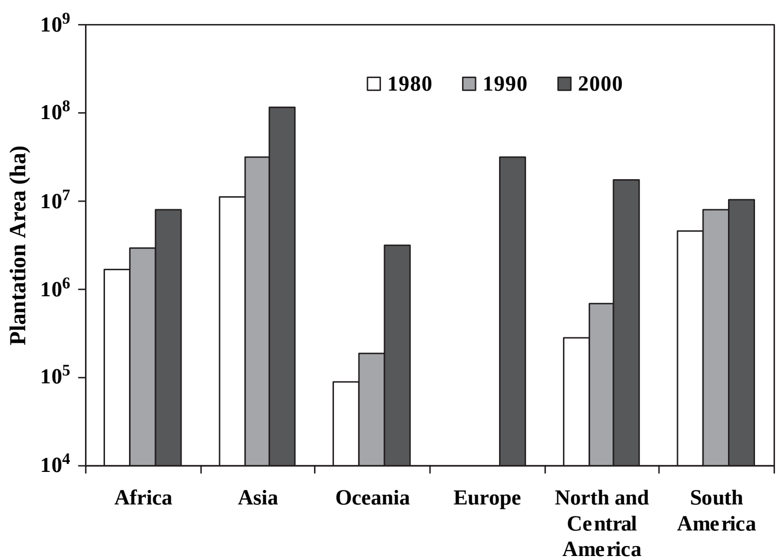

Fig. 12.12 Total area planted with trees from 1980 to 2000 for various continents (After FAO, 2000)

fiber, foods, and medicines but also protect soil and water resources. Reforestation has concentrated on planting *Pinus*, *Eucalyptus*, and *Acacia* species in tropical ecosystems. Planting a wide range of species is necessary to achieve simultaneous goals of soil and water conservation and production of food, wood, and bioenergy. Industrial monocultures with limited number of tree species have generated important benefits but not to the extent of ecosystem services provided by natural forests.

12.11 Afforestation

Afforestation refers to the process of planting trees in lands that previously did not support trees. These trees produce wood and fiber and conserve soil and water. Unlike reforestation, afforestation increases the total forest area by growing trees in pasturelands, rangelands, shrublands, and agricultural lands. In some ecosystems, trees can expand naturally if logging, grazing, and other human disturbances are curtailed. Planting trees in degraded or abandoned agricultural lands is the best approach to restoring degraded lands. It is a useful strategy to counteract or reverse the accelerated global deforestation.

Soil erosion. Afforestation is an effective approach to reduce soil erosion risks. While soil erosion hazard from recently afforested soils may be high due to soil disturbance, the risk rapidly decreases with increase in vegetation density. Planting trees in association with shrubs and grass-like vegetation to form multistorey canopies is a desirable strategy. A stand comprising mixture of different plant species is more effective in erosion control than those of single species. Runoff and soil erosion decrease exponentially with increase in vegetation cover or density, species richness, and diversity. In Spain, afforestation with Aleppo pine, a drought resistant species, is common in southern semiarid regions for recovering the vegetation cover and reducing soil erosion. Planting Aleppo pine trees on grasslands and shrublands reduced runoff volume by 10 times and soil erosion by 182 times (Chirino et al., 2006).

Soil properties. Planting trees improves the properties of degraded soils. Improvement in soil properties is often small in the first few years and then increases rapidly. An experiment in northern China showed that 3-, 9-, and 19-yr-old plantations of poplar gradually increased the soil organic matter content and plant available water capacity in sandy soils (Shirato et al., 2004). Gradual build-up of forest floor over time results in improved soil fertility.

12.12 Management of Cleared Forestlands

Restoration of biodiversity, increase in food supplying capacity, and improvement in environmental services provided by degraded forestland are an absolute necessity in a land-scarce world. The new paradigm is to undertake reforestation to achieve two simultaneous goals which are: (1) provision of ecological and environmental services, and (2) reduction of poverty of rural communities. The ideal approach to preserve the existing forestlands is to eliminate deforestation. This notion is, however, difficult to grasp in impoverished regions of the world where food production must be increased and farmers have no choice but to clear new lands for growing food crops. Thus, in regions where clearing is absolutely necessary, appropriate clearing methods and post-clearing management practices must be implemented. Because mechanized clearing with heavy equipment causes drastic soil disturbance, its use must be minimized.

Soil and water conservation practices must be installed following land clearing. Mulching, growing legumes, establishing grass fallows, using complex crop rotations, and adopting no-till are some of the recommended conservation practices. Adoption of no-till management in deforested lands is a useful strategy to restoring soil properties following deforestation. On a clayey soil in Brazil, soil structural development of a deforested soil increased with increase in duration of no-till cultivation and approached to that of a neighboring native forest site after 12 yr (De Assis and Lancas, 2005).

Management strategies of degraded forestlands differ among soil type, topography, and climate. Global overview of forest dynamics, degradation factors, removal of woods and non-wood products, and management are described by FAO (2005). Deforestation rates are specific to each continent and depend on the extent of remaining forest cover, biodiversity, and fragmentation. Afforestation alone may not be sufficient to curtail erosion in highly erodible soils. In such a case, combination of biological and mechanical measures is a viable alternative. The recommended practices for managing deforested lands are (Fig. 12.13):

- Avoid clearing steep slopes to reduce excessive runoff and soil erosion.
- Reduce clearing new areas by increasing the productivity of existing farmlands.

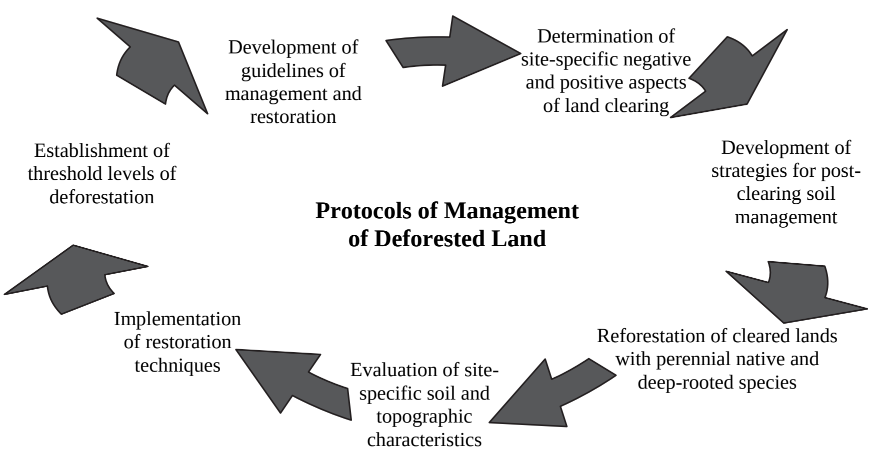

Fig. 12.13 Some specific steps to manage deforested lands

- Reduce the livestock stocking rates, intensive grazing, and trampling.
- Establish conservation buffers and mechanical erosion control practices.
- Establish perennial tree and grass species in rotation with row crops.
- Establish mixed cropping systems, legume cover crops, and crop rotations.
- Return crop residues, apply green and animal manure, and maintain permanent surface cover.
- Minimize tillage and adopt reduced tillage, no-till, mulch tillage, and strip tillage practices.

Reforestation is an effective practice to offset deforestation. Restoration of forest lands is possible, unless soil degradation has crossed the threshold levels with severe loss of topsoil and soil organic matter, causing irreversible consequences. In some regions (e.g., Europe) the trend of deforestation has been reversed between 2000 and 2005, and thus gains in total forest area have been reported due to enhanced afforestation and expansion of trees to agricultural and urban areas (FAO, 2005).

Despite advances in reforestation and establishment of protected forest areas across the world, the resulting modified forestlands are overly simplified and fail to capture the heterogeneity and biological richness of natural forest landscapes. Until now, reforestation and agricultural development of cleared forest lands have not been sufficient to provide all the benefits (e.g., wood and non-wood products, soil and water conservation) once provided by natural forests. Improved management techniques of secondary forests and complex forms of reforestation to restore productivity and biodiversity severely degraded and meager remaining forestlands must be adopted (Lamb et al., 2005). Important and pertinent questions including what species to plant, when to plant, where in the landscape to plant, how much land must be taken out for planting, and what are the economic returns of the plantations must be addressed.

12.13 Modeling of Erosion Under Forests

12.13.1 Empirical Models

The USLE technology is a simple empirical model for erosion prediction (Refer to Chapter 4) (Wischmeiter and Smith, 1978). Application of USLE technology to erosion prediction in forests has been, however, limited because USLE does not accurately capture the complex forest landscapes (e.g., steep slopes, rugged topography) (Elliot, 2004). Because the USLE is based on the principles of Hortonian flow (Wischmeiter and Smith, 1978), its utility for predicting erosion in undisturbed forest soils with high macroporosity and thus high saturated hydraulic is limited (Sheridan and Rosewell, 2003).

Contour maps of R-values have been modified in some forest soils (e.g., Australia) to improve the predictive ability of the USLE (Sheridan and Rosewell, 2003). The RUSLE is also used in combination with GIS and remote sensing to improve the accuracy of predictions. The GIS is used to gather and store soil survey data and generate soil maps, while remote sensing provides information on forest cover and digital elevations required to create cover-management factor, soil erodibility factor, and topographic maps for the RUSLE. This combined approach of RUSLE, GIS, and remote sensing has been recently used to map the spatial distribution of soil erosion patterns and severity across the Brazilian Amazon under intense deforestation (Lu et al., 2004). Despite its limitations, RUSLE is a valuable tool to predict soil erosion risks from conversion of forest to agricultural lands.

12.13.2 Process-Based Models

While sophisticated and process-based erosion models such as WEPP have been widely used in agricultural lands, their application to forestlands to predict runoff and soil erosion from forestlands has been slow and difficult (Covert et al., 2005). This is due in part to the requirement of a comprehensive and large input datasets for a successful application. The WEPP model was originally designed to predict soil erosion from relatively uniform agricultural watersheds and not from forestlands having high spatial complexity and variability in terrain, drainage, hydrologic, landscape, and vegetation characteristics. There is a greater distribution of sediment and runoff in forestlands due to the extensive network of undisturbed and disturbed (skid trails, roads) patches of land. Disturbed spots generate most of the runoff and sediment transport whereas undisturbed spots sink runoff sediment and reduce sediment yield, causing heterogeneities in sediment distribution within the forest. Surface and subsurface water flow including overland flow, interflow, lateral flow, baseflow, and percolation in forests differ from those in agricultural lands. For example, WEPP model assumes that the values of horizontal and vertical saturated hydraulic conductivity are the same, which may not be true for all soils especially in forest soils. Difficulties in simulating the complex vectorial components of water flow and rainfall partitioning undermine the successful application of the WEPP model in forests.

The WEPP model has, however, been modified and improved, and its database has been greatly expanded for forest erosion predictions (Refer to Chapter 4). Modifications to account for the forest soil hydrology and rainfall partitioning have resulted in improved predictions (Covert et al., 2005). The modified WEPP versions have been combined with GIS tools (GeoWEPP ArcX 2003) for better predictions across topographically and hydrologically complex forest ecosystems (Renschler, 2003). A new and large database accounting for the various sources of erosion in forestlands such as timber harvesting, road constructions, fires, log landings, and soil disturbance have been recently added into the WEPP Internet Interface through the work of U.S. Forest Service (Elliot, 2004). Three specific WEPP interfaces including *WEPP:Road* for predicting erosion from roads, *Disturbed WEPP* for predicting erosion from forest disturbance (e.g., timber harvesting, forest thinning, fires, wildfires, skid trails), and *Rock:Clime* for storing and providing all the database for the performance of the WEPP:Road and Disturbed WEPP have been incorporated (Elliot, 2004). This larger dataset compared to previous versions allows assessment of soil erosion across diverse climatic and forest conditions to produce predicted values close to measured values. These improved WEPP interfaces can also estimate erosion from shrublands and native grass prairies. The TOPOG is another physically-based model that partitions the land in topographic contours and flow lines across complex terrains (O'Loughlin, 1986). It is a digital terrain model that describes hydrological processes based on topographic features although it has been used less than WEPP.

Process-based models require further refinement for a variety of soil, topographic, and forest management conditions. Work on forest erosion has been mostly focused on measuring runoff and soil loss from each sediment source (roads, trials) under simulated rainfall at small scale. Hillslope models can permit the integration of all factors at larger scale. Remote sensing, satellite imagery, GIS, and computer predictive models are vital tools to monitor and study relationships among deforestation, soil degradation, water quality, tree growth, and changes in vegetative cover (Bakoariniaina et al., 2006).

Summary

Forests conserve soil and water, sequester C, mitigate net emissions of greenhouse gases, and provide wood and other products. Forests influence climate, conserve biodiversity, and improve environmental quality. Forestlands are classified in: primary, modified, semi-natural, and forest farms. Primary forests are natural forests and are little or no affected by anthropogenic activities while the rest of forest systems is, at some degree, managed or influenced by humans. Soil erosion from undisturbed forest soils is smaller than that from agricultural soils and ranges from 0.02 to 1.2 $\mathrm{Mg\,ha^{-1}}$. The low rates of soil erosion are explained by the dense canopy layers, thick floor litter, and extensive root system. The dense surface cover intercepts and sponges up raindrops and promotes water infiltration. It reduces wind velocity and thus wind erosion. Deforestation is a major threat to forests. Clear-cutting, slash-and-burn agriculture, excessive grazing and trampling, uncontrolled commercial logging, excessive tree harvesting, wildfires and uncontrolled fires, mining industries, and rapid urbanization are the main causes of deforestation. Agricultural expansion is the main cause of deforestation. About 13 Mha of forest are annually cleared worldwide and are converted to croplands and pasturelands.

Deforestation causes runoff and soil erosion, reduces C cycle, affects local and global climate change. Deforested soils are prone to rapid degradation. Seeding, mulching, and contour logging are among the strategies to manage burned forests. Prescribed forest fires are also used for controlling forest density and distribution and rejuvenating productivity.

Reforestation and afforestation are practices that counteract deforestation. Degradation of deforested lands can be reduced by avoiding clearing steep slopes, reducing livestock stocking rates, excessive grazing, and trampling, and establish conservation buffers, growing perennial tree and grass species, establishing legume cover crops and crop rotations, mulching with crop residues, and applying organic amendments, and adopting reduced tillage and no-tillage practices. Models, remote sensing, geographic positioning systems, and geographic information systems are improved technological tools to assess and manage forest cover and reduce deforestation.

Study Questions

1. Discuss various types of forests and their role in soil and water conservation.
2. Describe procedures to estimate runoff and soil erosion rates for natural, modified forests, and forest farms.
3. Discuss the causes of deforestation and soil degradation.
4. Discuss the mechanisms by which hydrophobicity of soil reduces water infiltration and increases runoff and soil erosion.
5. Explain how modeling tools can be used to reduce deforestation. Show examples.
6. Discuss differences in forest management across contrasting climate regions (drylands, temperate, and tropics)
7. Describe the strategies for forest management.
8. Discuss the differences between afforestation and reforestation.
9. Explain how prescribed forest fires contribute to forest management.
10. Why are forest soils more water repellent than cultivated soils?

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Chapter 13
Erosion on Grazing Lands

Grazing land is defined as a land unit consisting mostly of grass and herbage where animals graze (Fig. 13.1). In this Chapter, pasturelands, grasslands, meadows, and rangelands are all considered an integral component of grazing lands. These ecosystems differ in their management, plant species composition, and distribution, and are of vital importance to soil and water conservation. Depending on their life cycle duration, plant species used for grazing can be *annuals* if they complete their life cycle in ≤ 1 yr, *biannuals* if they complete in two years, and *perennials* if they live ≥ 2 yr. Annuals often reproduce from seeds and can grow at different times during the year (Guretzky et al., 2005). For example, summer annuals emerge in spring and die before winter, whereas winter annuals sprout in fall and complete their life cycle in summer of the next year. Biannuals are not as common as annuals or perennials. They develop their root system in the first year, and produce seeds in the second.

Fig. 13.1 Localized concentration of animals results in overgrazing and increases risks of soil degradation (Courtesy USDA-NRCS)

Perennials consist of sod- or bunch-type grasses, which grow vigorously in spring and remain dormant in winter.

Based on their response to grazing and environment, grasses are *decreasers* if they are adversely affected by excessive grazing, *increasers* if their production increases after grazing, and *invaders* if the vegetation has no forage value and impedes proliferation of high value forages (Dyksterhuis, 1949). Richness in plant species under grazing depends on the grazing intensity, ecosystem, and climate.

The grazing lands can be grouped into two main categories: *rangelands* and *pasturelands*.

13.1 Rangeland Systems

Rangelands may consist of natural grasslands, pasturelands, shrublands, meadows, tundras, coastal marshes, and savannas. These are complex and diverse ecosystems predominated by native grasses, grass-like vegetation, forbs, and shrubs growing in either natural or recreated lands under different scenarios of grazing and management. Rangelands are more complex than pasturelands and comprise a host of natural systems (e.g., woodlands, savanna). These lands are often not suited for cropping and forest management. In the western U.S., the region predominated by range farming, rangelands are mainly comprised of pasturelands with scattered shrubs and trees. Unlike urban or agricultural lands, rangelands do not refer to a land use, rather to a type of land.

Rangelands have multi-purpose uses including animal grazing, firewood production, recreation, landscape scenery, wildlife habitat, and ecotourism. While, traditionally, rangelands have been primarily used for livestock production, there is now an increasing recognition about their importance to wildlife habitat, soil hydrology, water quality, tourism, and contemporary ecosystem services.

13.2 Pastureland Systems

Pasturelands consist of single- or native multi-grass species and grass-legume mixtures. Pasturelands that consist mostly of grasses are known as “grasslands”, which occupy one of the largest ecosystems in the world covering about 40% of the terrestrial ecosystem (Fig. 13.2; WRI, 2000). These are distributed across a wide range of ecosystems differing in plant species and grazing pressures. Natural pasturelands are rich in flora with about 10,000 grass species worldwide although only about 150 species are cultivated. Pasturelands fit between desert and forest land and extend across landscapes with sufficient water to grow grasses. The most extensive grazing lands are in northern China; mountains of Himalaya Hindu Kush in South Asia; the Great Plains in the USA; Las Pampas, Cerrados,

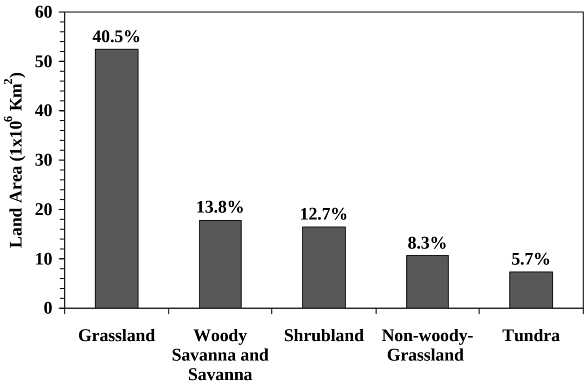

Fig. 13.2 Grassland surface area and percent of the total terrestrial area in the world (After WRI, 2000)

the Chaco, and Altiplano in South America, Australia, and eastern Africa (Suttie et al., 2005).

Grasslands and *meadows* are considered part of pasturelands.

- ***Grasslands*** are lands predominantly covered by grass vegetation with $<$10% tree and shrub cover, while *wooded grasslands* include 10 to 40% tree and shrub cover (Suttie et al., 2005). The production and management of grasses, legumes, and fodder are referred to as *a grassland agriculture*, which is a system where plants, animals, and soil and water resources are managed interdependently. The goal of grassland agriculture is not only to produce forage for livestock production but also to incorporate strategies for soil and water conservation and management.
- ***Meadows*** are natural systems for grazing and are generally composed of native species, similar to grasslands and tundra. Wet meadows remain moderately wet both in winter and in summer. Meadows are tracts of grassland in which native and introduced plant species differ in their characteristics from those of surrounding landscape because of differences in landscape position, hydrology, erosion hazard, and soil conditions. According to their environment and use, meadows can be *hay*, *native*, *mountain*, *lowland*, and *wet* meadows. Hay meadows are usually established on well drained soils for ease of hay harvesting.

Pasturelands provide a set of vital ecosystem services to livestock production, plant biodiversity, wildlife habitat, soil and water conservation, recreation, tourism, conservation of plant genetic resources, and environmental quality. Dense grass species stabilize soil, reduce soil erosion, and moderate fluxes of water, heat, and gas. Grass cover is the most environmentally friendly and economically attainable measure to protect soil from erosion. Pasturelands play a major role in improving the environmental quality by reducing water and wind erosion, sinking pollutants, sequestering C, and buffering the whole ecosystem.

13.3 Degradation of Grazing Lands

13.3.1 Rangelands

The ever-increasing population pressure on rangelands has exacerbated desertification of these ecosystems at the global scale. Desertification refers to the process of becoming desert due to land use and management or/and climate change. While drought is a major cause of desertification of rangelands, other environmental disasters (e.g., extreme weather events) in association with increase in animal stockings, grazing, and harvesting of sparse trees exacerbate the problem. Desertification can occur not only in dry areas but also in any ecosystem subject to large variations in climate, such as Iceland (Arnalds, 2000). Excessive grazing disturbs at small scale, but its interaction with prolonged drought periods and intensive tree clearing causes desertification.

Cattle grazing in many countries adversely affects soil attributes, biomass production, and litter amount. Rangelands in dry regions with sparse vegetation (e.g., shrubs) are more prone to desertification than those (e.g, savannas) in humid and tropical climates. Rate of desertification is a function of the level of perturbation/stress and the resilience/stability of the ecosystem. Any ecosystem is subject to continuous small disturbances, but stresses above critical levels lead to desertification and instability of the system. Urbanization is one of the contemporary threats to rangeland management because it adversely affects wildlife habitat, impairs water quality, and reduces rangeland area. Extensive rangeland areas have been turned into urban areas and fragmented with road networks and other invasive activities. Rangeland area affected by degradation in the USA is described by Lal et al. (2004).

13.3.2 Pasturelands

Pasturelands are, at present, are seldom natural ecosystems in that they are strongly influenced by human activities such as *intensive cultivation, excessive grazing, fires, road constructions, and introduction of invasive plant species*. While the extent of pasturelands is still large, their capacity to provide all the essential services is declining due to rapid degradation and fragmentation. Conversion to agricultural and urban land uses is the main human-induced factor that is reducing the total area of pasturelands (Fig. 13.3). The world's most productive natural grasslands (e.g., North American Prairie, Cerrados in Brazil, the South American Pampas, the East European Steppe) are being gradually converted to prime agricultural lands, and grazing is being relegated to fragmented and marginal lands. In Africa, lands which receive sufficient rainfall have been converted to intensive cultivation, and meager grasslands are being integrated with cultivated lands for subsistence farming. Burning of grass and shrubs in Africa is a major cause of soil degradation. Scarcity of water is also a major determinant of pasture development in dry regions. Excessive grazing in arid, stony, shallow, flooded, mountainous and remote areas has exploited

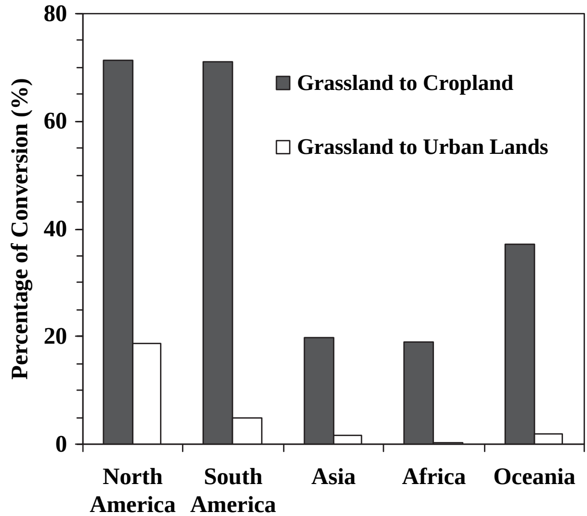

Fig. 13.3 Percentage of grassland converted to cropland and urban areas around the world (After White et al., 2000)

natural pasturelands. The idea that “grass is for grazing” must be balanced against the vital role of grasses for conserving soil and water and sustaining forage and animal production.

The reduction in pastureland areas due to population growth has led to an increase in soil erosion risks, off-site transport of pollutants, and decline in diversity of flora and fauna. Ecologically complex but fragile rangelands are gradually being converted to simpler and somewhat artificial pasturelands (e.g., seeded vegetation). In the USA, the Great Plains contain the most extensive grassland areas, but about 70% of these lands have been fragmented in blocks of $<1{,}000\,km^2$ in size by human intervention (White et al., 2000). About half of the beef cattle in the USA are concentrated in the Great Plains, predominating over sheep. In the USA and Europe, about 50% of pasturelands are degraded with 5% prone to severe degradation (White et al., 2000). Tallgrass prairie in the USA and savanna in Brazil (Cerrados), Paraguay, and Bolivia are important but declining natural reserves of pasturelands. Introduction of non-native or exotic plant species is another factor that has reduced biodiversity due to the higher competition from introduced species. In terms of soil and water conservation, introduced short growing grass species (e.g., tall fescue) are less beneficial than native species because these have more extensive and deeper root systems and grow taller. About 10 to 20% of grasses in the Great Plains are non-native (White et al., 2000). The extent of pastureland degradation in the USA is described by Lal et al. (2004).

Magnitude and rate of degradation of pasturelands vary around the globe due to differences in local and regional conditions such as:

Management. There are two types of grazing systems: commercial and traditional. The former refers to large-scale production of livestock (e.g., cattle, sheep) normally grazed on intensively managed pasturelands (e.g., seeding, use of fertilizers). Sown pasture is widely used in rotation with annual crops in commercial management systems. The latter is the traditional type of grazing and is practiced mostly for subsistence.

Climate. Climate influences vegetation density. Unlike mesic or temperate ecosystems where soils are deep and high in soil organic matter content, arid environment is extremely susceptible to excessive grazing. In arid and semiarid regions of the world, about 80% of the pasturelands are affected by some degree of degradation. Grazing in arid ecosystems decreases density of sparse vegetation and increases soil erosion by wind. Periodic droughts in interaction with excessive grazing cause rapid changes in grass composition. Degraded pasturelands in arid regions are less resilient due to reduced biological activity. In eastern Africa, extensive pasturelands occur under arid and semiarid climates with high incidence of drought and excessive grazing. Sloping pasturelands are more susceptible to erosion.

13.4 Grazing Impacts

There are differences between pasturelands and rangelands. Pasturelands are managed and receive high agronomic inputs (e.g., fertilizers, irrigation, manure) for forage production. Surplus forage production is harvested as hay. Pasturelands support high livestock stockings and are thus highly susceptible to soil erosion. In comparison, rangelands are often not cultivated and animal stockings rates are low relative to pasturelands. Both rangelands and pasturelands are, however, disturbed by grazing, fires, urbanization, and other human-induced disturbances.

Two among the many factors influencing the magnitude of soil erosion in pasturelands and rangelands are the *grazing- and trampling-induced reductions* of soil surface vegetative cover and *degradation of soil surface properties*. Combinations of both reduced vegetative cover and degraded soil surface exacerbate runoff and soil erosion.

13.4.1 Soil Erosion

Excessive grazing or overgrazing alters the natural grazing lands and increases accelerated runoff and soil losses by:

- removing biomass and reducing the vegetative cover and surface litter which: (1) leaves the soil bare, (2) increases the direct impact of raindrops, and (3) accentuates soil detachment, splash, and soil transport. Height of grass and amount of surface litter generally decrease with increase in grazing intensity,
- curtailing the root development and regrowth of plants and reducing biomass production.
- altering vegetation structure and distribution of flora and fauna and creating conditions for eventual desertification, and
- trampling or hoof action of animals which causes lateral displacement of soil, contributing to severe erosion.

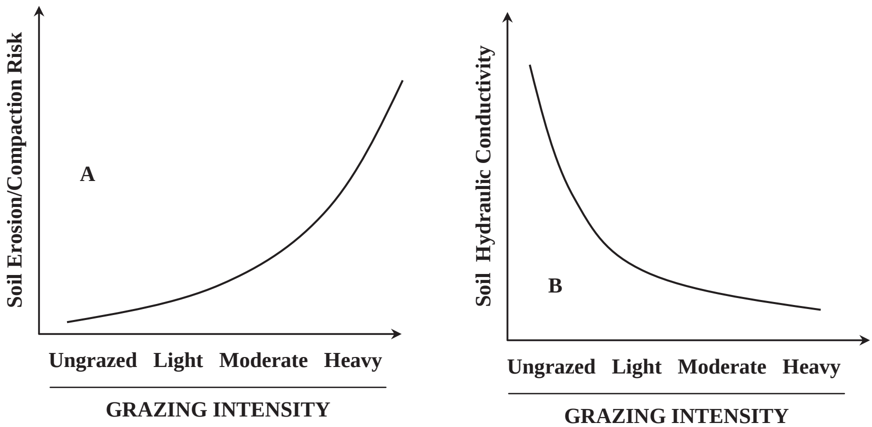

Fig. 13.4 (**A**) Runoff and soil erosion increase whereas (**B**) saturated and unsaturated water flow decreases with increase in grazing intensity

Rates of runoff, sediment, and nutrient loss are indeed negatively correlated with vegetative cover. Runoff, soil erosion, and compaction increase in direct proportion to reduction in vegetative cover (Fig. 13.4A). Changes in vegetation cover in pasturelands can explain as much as 80% of the variation in runoff. Dense (>70%) and diverse perennial grasses can reduce runoff to negligible levels (Descheemaeker et al., 2006). Amounts of runoff and soil erosion loss are the largest from heavily overgrazed soils and decrease with increase in vegetative cover due to reduced soil detachment.

Surface soil and landscape characteristics are important determinants of runoff and soil erosion. Overgrazed steep soils (>5%) are more susceptible to runoff and soil erosion than lowland pastures. During rainy seasons, animals tend to concentrate on the upper landscape positions increasing risks of soil erosion. In Argentina, pasturelands with short-growing grasses had higher rates of runoff or soil loss as compared to those with tallgrasses (Aguilera et al., 2003). Runoff from soils under continuous grazing is higher than from those under rotational systems. On steep terrains (18 to 37% slope) in Canada, snow melt runoff volume from intensively grazed watersheds under tall fescue (255 m^3) was higher by a factor of 118 compared to that from ungrazed watersheds (2.16 m^3) over 11 runoff events (Chanasyk et al., 2003).

The impact of overgrazing is even larger in desert or semidesert pasturelands with ephemeral grasses and scattered shrubs. In arid regions, trampling breaks soil crusts and aggregates and pulverizes the soil surface, causing the loss of fine particles through wind erosion. In semiarid regions in northern Mexico and southwestern USA, overgrazing has caused the replacement of pasturelands by shrublands (Ludwig and Tongway, 2002). While shrubs are important components of many pasturelands (e.g., savannas) for providing shade and shelter, excessive invasion and replacement of grass alter the ecosystem and increase runoff and soil erosion. In southern New Mexico, semiarid shrublands had greater losses (50.5 vs. 8.5 mm) of runoff than neighboring pasturelands (Schlesinger et al., 2000). There are numerous other examples of excessive grazing and its adverse effects on accelerated soil erosion worldwide. In steep lands of the Andes (>3000 m.a.s.l.) of Latin America, natural pastureland ecosystems are being gradually degraded by excessive grazing due to the increase in rural population and number of animals (e.g., sheep) (Podwojewski et al., 2002). The available land for grazing is progressively being reduced and is insufficient to satisfy the increasing demand for the growing population. Magnitude of grazing impacts depends on vegetation structure and resilience of the system. In some ecosystems while grazing alters the composition and structure of the vegetation, it does not always increase the population of invasive grass species (Altesor et al., 2006).

13.4.2 Soil Properties

Excessive grazing adversely impacts vegetation growth and soil physical, chemical, and biological properties. Frequent trampling or animal traffic loosens, homogenizes, and pulverizes soil surface, degrading near-surface soil properties. In unfenced pastures, trampling is not localized but often distributed across the whole field unlike field machinery traffic. Trampling increases soil bulk density and reduces macroporosity, water infiltration rate, hydraulic conductivity, aeration, and drainage. Converting native prairie to pasture is another factor that degrades soil physical quality. In some ecosystems, changes in soil properties from the conversion of native prairie to pasture or hay meadows can be small due to the high resilience of grazed systems.

13.4.2.1 Soil Water and Temperature Regimes

The surface of surface vegetative cover (canopy cover) moderates soil water content and temperature regimes. Grazing reduces the vegetative cover and increases percent of the bare soil area. Overgrazed systems with patches of bare soils have lower water content and higher daytime temperatures than grass-or litter-covered soils. Bare soils warm up and cool faster than those with a dense vegetative cover. Overgrazed soils typically undergo higher fluctuations in surface temperature. Overgrazing increases evapotranspiration and modifies soil microclimate. Alteration of soil water content and temperature regimes influences soil respiration, microbial processes, and other dynamic soil processes.

13.4.2.2 Soil Textural Characteristics

The hoof action of animals breaks aggregates, detaches primary soil particles, pulverizes the soil, and increases soil's susceptibility to wind and water erosion. Detached small soil particles are easily transported by wind and water erosion. As a result, heavily grazed soils have higher sand content and lower clay content (Table 13.1). In some excessively grazed soils, loss of topsoil by erosion may expose

Table 13.1 Impacts of overgrazing on selected soil properties

Soil property	Ungrazed	Grazed
Sand content (g kg^{-1})	716a	839b
Silt content (g kg^{-1})	221a	128b
Clay content (g kg^{-1})	63a	32b
Mean weight diameter (mm)	17a	14b
Water infiltration (cm day^{-1})	285a	50b
Organic C (Mg ha^{-1})	50a	15b

Source: Neff et al. (2005), Daniel et al. (2002), and Dormaar and Willms (1998).

subsoil layers with high clay content (e.g., claypan soils), which modifies the original surface soil texture.

13.4.2.3 Soil Structure

Excessive grazing in conjunction with the trampling effect degrades soil structure thereby reducing aggregate stability, pore-size distribution, macroporosity, total porosity, and water infiltration rate (Table 13.1). It reduces the amount of vegetative cover required to protect the exposed soil surface, increases surface sealing and crusting, reduces aggregation, and seals the open-ended macropores. Mixing and remoulding of wet soils are primary causes of deterioration of strength and stability of soil aggregates. In contrast, soil aggregates formed under grass are more stable and less detachable than those under bare soils. Well-aggregated soils are porous and have high water infiltration rates, but their perturbation increases the soil erodibility.

13.4.2.4 Soil Compaction Parameters

Compaction, consolidation, and puddling are the main processes set in motion by excessive grazing thereby altering soil physical quality. *Compaction* refers to the compression of unsaturated soils, whereas *consolidation* refers to the compression of saturated soils (Drewry, 2006). P*uddling*, which refers to soil deformation by plastic flow when perturbed under saturated conditions, is severe in paddocks with wet soils (e.g., spring) and large livestock concentration. Wet soils have low strength and are rapidly pugged by grazing animals. The magnitude of compaction depends on the animal weight. Cattle may compact soil more than sheep due to differences in body mass, but sheep hoof action often pulverizes dry soil surface and disturbs and mixes wet soil more than cattle. Normal stocking rates, shallow animal traffic, and use of rotational systems reduce compaction. The high soil organic matter content and abundant root biomass of pasturelands buffer against the compactive forces.

13.4.2.5 Soil Hydraulic Parameters

Decrease in surface vegetative cover and increase in trampling change the pattern of water flux. High levels of soil compaction restrict water and air movement in the soil. It reduces saturated and near-saturated hydraulic conductivities as formation of macropores (e.g., earthworm burrows, root channels) is reduced (Fig. 13.4B). Impacts of pugging are severe in clayey soils with limited infiltration. Excessive grazing reduces water infiltration and water retention capacity of the soil (Table 13.1).

13.4.2.6 Soil Organic Matter Content

Excessive grazing reduces C biomass input by removing portions of aboveground biomass. Thus, overgrazed soils have lower C pool than ungrazed systems (Table 13.1), and are more erodible and less fertile. The soil organic matter content provides essential nutrients, binds soil particles into aggregates, promotes soil structural development, and enhances microbial activity. While excessive grazing depletes soil organic matter content especially in arid and semiarid regions, moderate grazing can promote plant growth, enhance species diversity, and thus increase soil organic matter content.

13.4.3 Plant Growth

Excessive grazing and trampling degrade structure of the vegetative cover structure and alters its dynamics by directly altering the plant morphological characteristics. It reduces height and density of plants and amount of coarse litter cover (Fig. 13.5). While moderate grazing is beneficial to plant development, excessive grazing jeopardizes plant regrowth and reduces yields of hay or forage. It progressively degrades vegetative cover and its effects are greater on annuals than on perennials due to differences in biomass quantity and plant resilience. Denudation and sharp discontinuities in vegetation are typical in heavily grazed hilly terrains. Reduction in pasture production and plant diversity reduces livestock production. Damaged plants can not recover quickly and are often replaced by poorer species. Soils and vegetation of the arid and semiarid regions are the most susceptible to desertification by overgrazing.

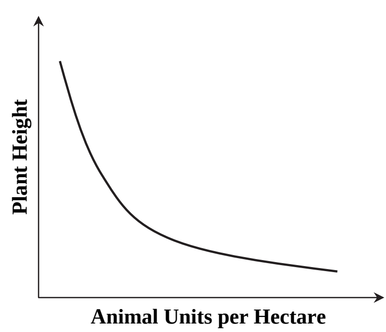

Fig. 13.5 Direct effects of grazing on vegetative cover

13.5 Grasses and Erosion Reduction: Mechanisms

Well-managed pasturelands and rangelands which have dense stands, thick surface litter, and abundant network of fibrous roots can reduce splash by impacting raindrops, increase rain infiltration, and thereby reduce runoff and soil erosion. Growing grasses reduce runoff and soil erosion by two interrelated mechanisms, which are *protection* of the soil surface by the vegetative cover and *stabilization* of soil matrix.

13.5.1 Protection of the Soil Surface

The above-ground biomass intercepts raindrops and incoming runoff, reduces the velocity and transport capacity of runoff, spreads and ponds runoff, and increases water infiltration opportunity time. Grass density, stem stiffness, and height are important factors which determine the resistance vegetation to erosion. The Manning's coefficient of roughness increases and turbulent runoff flow or Reynolds's number decreases with increase in vegetative cover. It also varies depending on the grass type. The roughness coefficient varies with each individual grass species and ranges from 0.02 to 0.4. Vegetation roughness increases linearly with grass cover regardless of changes in soil slope (Mishra et al., 2006). The lower the roughness coefficient value, the lower is the ability of grass to resist runoff flow and soil erosion.

The root system is as important as, if not more than, the above-ground biomass to controlling soil erosion. While the surface biomass protects the soil from the erosive impact of raindrops, the roots hold the soil in place. In overgrazed systems, grass roots play an important role in reducing soil erosion as the effectiveness of above-ground biomass cover for controlling soil erosion is diminished by grazing. Much attention has been paid to the effects of surface vegetative cover on soil erosion while the vital functions of below-ground biomass in soil erosion have been somewhat neglected. Grass roots interact with the soil matrix, stabilize sloping soils, and reinforce the soil's resistance against slippage.

13.5.2 Stabilization of Soil Matrix

The below-ground biomass (e.g., roots) in interaction with soil organisms stabilizes the soil, enhances biological activity, and increases water infiltration rates, thereby reducing soil erodibility. Thus, erosion reduction and improvement in soil properties are the result of the combined effects of the increase in the above- and below-ground biomass. Management of above-ground biomass is crucial to maintaining a dense and stable root system. Living roots control the soil erosion better than do dead roots. The dense network and cohesiveness of roots stabilize the system and prevent scouring and development of head-cuts when concentrated runoff occurs.

Overgrazed and short-growing vegetation have lower hydraulic resistance to concentrated overland flow than tall and stiff-stemmed grasses. When the above-ground biomass is overcome by accelerated erosion (e.g., concentrated flow) grass roots become major constituents of the system against the formation of rills and ephemeral gullies. The network of fine and coarse roots also penetrates deep into the soil, open up compact layers, increase saturated and unsaturated water flow, and reinforce the soil structural strength, enhancing the soil's resistance against rill and gully erosion.

Most grasses have fibrous and fine roots with diameters ranging between 0.15 and 0.24 mm, forming a dense natural mesh. Roots, especially if abundant and fibrous, add organic matter to the soil, bind soil particles, promote aggregation, and enmesh aggregates, reducing their breakdown. They promote assemblage of soil particles into stable aggregates through physical, biological, and chemical bonding mechanisms, as follows:

1. ***Physical binding*** refers to the mechanical enmeshment of soil particles by roots.
2. ***Biological binding*** occurs when root-derived organic binding agents such as exudates and glue-like compounds infiltrate and stick the soil particles together.
3. ***Chemical binding*** refers to the reaction of root-derived substances along with inorganic compounds.

Grass roots either live or dead reduce soil erosion by:

1. *increasing water infiltration, permeability, and drainage*,
2. *loosening compact layers (e.g., hardpan, claypan, and plowpan)*,
3. *enmeshing soil particles and form stable aggregates*,
4. *developing biochannels and increase soil macroporosity, and*
5. *increasing soil organic matter content and microbial processes*.

Roots of grasses are more effective than those of common crops because roots under natural vegetation remain less disturbed and penetrate deeper into the subsoil. These beneficial processes of growing grasses depend, however, on the management of pastureland. Intrinsic characteristics of roots (e.g., diameter, density, hair, length, age) control aggregation. Distribution and interaction of roots within the soil matrix govern the strength and stability of aggregates. The higher the density and longer the roots, the lower the aggregate detachment and slaking. Soils under grass can have detachment rates of 30–90% lower than those under bare soils.

13.6 Root System and Soil Erodibility

Soil erosion decreases exponentially with increase in grass root density. Several empirical equations have been developed relating grass root density and length to soil erodibility (Alberts et al., 1995; Mamo and Bubenzer, 2001; De Baets et al., 2006). Some of these relations are listed below:

$$Y = a \times \exp(k \times RD) \tag{13.1}$$

$$Kr_{lr} = \exp(-3.5 \times lr) \tag{13.2}$$

$$Kr_{dr} = \exp(-2.2 \times dr) \tag{13.3}$$

$$RSD = \exp(-1.14 RD) \tag{13.4}$$

$$RSD = \exp(-0.0062 RLD) \tag{13.5}$$

$$RSD = \frac{RD^{-1.76}}{1.59 + RD^{-1.76}} \tag{13.6}$$

$$Kr = 0.0017 + 0.0024 \times clay - 0.0088 \times OM - \frac{0.00088 \times \rho_b}{1000} - 0.00048 \times root \tag{13.7}$$

$$Kr = 42.66 \times \exp(-0.323 \times RLD) \tag{13.8}$$

$$K_i = 3.55 \exp(-0.71 \times RTM) \tag{13.9}$$

$$K_i = 3.62 \exp(-0.029 \times RTL) \tag{13.10}$$

where *Y is s*oil erodibility, *a* and *k* are constants, *RD* is root density (kg m^{-3}), *RSD* is relative soil detachment rate (0–1), *RLD* is root length density (km m^{-3}), *Kr* is rill erodibility (s m^{-1}), *clay* is clay content (0–1), *OM* is soil organic matter content (0–1), ρ_b is soil bulk density (kg m^{-3}), *lr* is mass (kg m^{-2}) of living roots, and *dr* is mass (kg m^{-2}) of dead roots, root is total root mass (kg m^{-2}), K_i is interrill erodibility parameter of the soil (kg s m^{-4}), *RTM* is dead root mass (kg m^{-2}), and *RTL* is dead root length (kg m^{-2}). The high number of such empirical relations is partly attributed to their site- and root-specific influences on erodibility. Thus, these equations must be adapted in other regions only after proper local validation.

Understanding the critical shear stress levels of vegetation in relation to onset of interrill and rill erosion is vital to sustainable management of pasturelands. Standing tall grass provides higher friction and greater hydraulic resistance to runoff than submerged stands. Reduction of vegetation height and stem density by grazing results in reduced friction and increased mean flow velocity. Formation of gullies depends directly on the critical shear stress of the grass roots, whereas interrill flow depends mostly on soil surface conditions and above-ground biomass cover. Most short-growing and overgrazed grasses bend over and fail under concentrated flow with high flow velocity. Vegetation length and stiffness are two important parameters that determine the resistance of the grass media to concentrated runoff.

Flow velocities (V_m) (m s^{-1}) in concentrated runoff flow in pastureland depressions are estimated using the modified form of Manning's equation as follows (Haan et al., 1994):

$$V_m = \left(\frac{1.0}{xn}\right) R_s^{2/3} S_c^{1/2} \tag{13.11}$$

where xn is adjusted form of roughness coefficient, R_s is hydraulic radius of the grass channel (cm), and S_c is slope of channel (cm cm^{-1}). The Darcy-Weisbach friction factor is used to determine the flexible roughness of grass stems where the friction factor (f) is computed as per Eq. (13.12) (Kouwen and Li, 1981)

$$f = \frac{8gd_f Sc}{{V_m}^2} \tag{13.12}$$

In terms of relative submergence of grass, Eq. (13.12) is transformed into Eq. (13.13) and (13.14):

$$\frac{1}{\sqrt{f}} = a + b \log\left(\frac{d_f}{k}\right) \tag{13.13}$$

$$k = 0.14h \left(\frac{\left(\frac{MEI}{\rho_w d_f S_c}\right)^{0.25}}{h} \right)^{1.59} \tag{13.14}$$

where g is gravitational constant (cm s^{-2}), d_f is runoff flow depth (cm), a and b are fitted parameters which depend on the grass roughness and channel geometry, h is height of grass (cm), ρ_w is density of water (g cm^{-3}), M is density resulting from the ratio of the grass stem count per unit area to a control/reference stem density in which the reference number is often taken equal to 1, E is modulus of elasticity (Pa), and I is second moment of the cross-sectional area of the stems. The I is estimated using Eq. (13.15) assuming that the stems are circular (Dunn and Dabney, 1996)

$$I = \frac{\pi}{64}\left({d_1}^4 - {d_2}^4\right) \tag{13.15}$$

where d_1 is outside diameter (cm) of the stem and d_2 is inside diameter (cm) of the hollow of the stem. The MEI product refers to the rigidity of the stems (N m^{-2}). The stiffness of vegetation can be estimated by determining the shear velocity of runoff and the critical shear velocity of grass. The critical shear of grass is predicted as (Kouwen and Li, 1981; Haan et al., 1994) per Eqs. (13.16 and 13.17)

$$V_{crit1} = 0.028 + 6.33(MEI)^2 \tag{13.16}$$

$$V_{crit2} = 0.23(MEI)^{0.106} \tag{13.17}$$

where V_{crit1} is critical shear velocity of elastic grass and V_{crit2} is critical shear velocity of stiff grass. Values of MEI for common grasses are estimated as per Eqs. (13.18 and 13.19) (Kouwen, 1992)

$$\text{Growing Grass}: MEI = 319h^{3.3} \tag{13.18}$$

$$\text{Dormant Grass}: MEI = 24.5h^{2.26} \tag{13.19}$$

The actual shear velocity (V^*) is computed as (Kouwen and Li, 1981)

$$V^* = \sqrt{gd_f Sc} \tag{13.20}$$

whenever the V^* value exceeds either the V_{crit1} or V_{crit2} values, the vegetation fails under concentrated runoff. Additional approaches to estimate the sediment trapping efficiency and filtering capacity of different grass species are described in detail by Haan et al. (1994).

Grass roots change the shear strength of the soil matrix. Interaction between the high tensile strength of roots and low shear strength of soil results in higher effective shear and tensile strengths. Their effect on shear and tensile strength depends on grass density, strength, modulus of rupture, elasticity, length, diameter, orientation, penetration depth, and age. Shear strength in soils containing roots of switchgrass can be about 5 times higher than that in soils without grass roots (Goldsmith, 2006). Soils under grass commonly have higher shear strength than those under shrubs because of higher density of fine grass roots. Soil shear strength in pasturelands is higher compared to that in shrublands due to the dense and abundant roots under pasture.

13.7 Water Pollution in Grazing Lands

While pasturelands are vital for forage and livestock production and soil erosion control, their mismanagement (e.g., overgrazing, excessive trampling) can, however, cause severe soil degradation and decline in water quality. Soil surface disturbance and dung deposition are sources of pollutants such as sediment and nutrients, which impair water quality. Concentration of sediment and nutrients in runoff increases with excessive grazing (Elliott and Carlson, 2004). The combination of reduced water infiltration, high rainfall rates, and short grazed pastures increases transport of pollutants. Runoff and pollutant transport gradually decrease with cessation of grazing as the grazed pastures recover over time. Unrestricted grazing is the major source of pollutants. Streambank erosion is another common cause of water pollution in pasturelands.

Well-maintained pasturelands and rangelands buffer transport of pollutants. Their effectiveness for filtering sediment, nutrients and pesticides depends on the density of stems and patterns of root system. Assessment of total maximum daily load (kg d^{-1}) (TMDL) of sediment, nutrients, and toxic chemicals is required for each grazing land. Long-term pollutant load estimations from watersheds and simulation models, and remote sensing techniques provide information of current and future trends in soil erosion and water quality. Long-term and abundant records of daily, seasonal, and annual changes in upland land use, streamflows, water quality, and other soil inherent data are important to estimate sediment and chemical loading rates and design measures for controlling pollutant loads.

13.8 Grazing and Conservation Buffers

Filter strips and grass barriers are useful conservation measures to reducing transport of pollutants to streams, rivers, and lakes. Buffers placed downstream of agricultural lands filter, sink, and trap sediment and degrade sediment-bound chemicals (Refer to Chapter 9). The effectiveness of grass buffers in reducing runoff and soil loses is a function of height, density, and age of vegetation and characteristics of upland soils. Excessive grazing and the attendant animal trampling reduce the height and density of grass species, the critical shear stress, hydraulic resistance to runoff, and sediment filtering capacity of grasses. Increased runoff velocity causes inundation of short grasses and overtopping, which dramatically reduces grass effectiveness for trapping sediment. In a riparian meadow in northern Colorado, cattle grazing reduced grass stem density by 40%, the aboveground biomass by 61%, and the sediment trapping efficiency by 13% (Mceldowney et al., 2002). Reduction of grass buffer density decreases the tortuosity of rills or small channels and induces formation of concentrated flow channels. Grass buffers are particularly effective at trapping coarse soil particles as fine particles or colloids remain suspended in runoff water. Excessively grazed buffers lose, however, their ability to trap coarse particles. Mceldowney et al. (2002) reported that sand transport increased by about 80% when buffers are grazed. Overgrazing is more damaging than trampling because it removes vegetation. Rough surface intercepts and ponds runoff water and promotes infiltration. Decrease in surface cover below 50% dramatically increases soil losses. Vegetation height is important but more crucial than is the stem density which increases roughness in association with surface residues. High density of stems provides greater hydraulic resistance to incoming runoff and sediment delivery.

In the USA, the USDA-NRCS through the National Conservation Buffer Initiative set out in 1997 to help farmers or landowners establish 3.2 million km of conservation buffers (e.g., riparian buffers, vegetative filter strips, grass barriers, grassed waterways) by 2002 (USDA-NRCS, 1999). The goal of this massive establishment of buffers was to reduce non-point source pollution of water resources, improve air quality, promote wildlife habitat diversity, and foster the beauty of green landscapes. Although the goal of establishing 3.2 million km was not completely met by the end of 2002, notable progress was made towards achieving the goal, and about two-thirds of USDA-NRCS's goal of 3.2 million km of conservation buffers was achieved. A continued expansion of buffers is required to reduce the ongoing concerns of water pollution with non-point source pollutants. Buffers installed within, at the edge, and below crop fields have proven to be an effective technology to offset pollution.

13.9 Grasslands and Biofuel Production

Grasses are not only used for animal grazing and for providing numerous ecosystem services but also possibly for biofuel production as a source of lignocellulosic biomass. There is an increasing interest in the importance of mixed perennial grasses for energy production due to increased fossil fuel costs and environmental pollution including the accelerated greenhouse effect. Interest in growing grasses as a source of biofuel feedstock is particularly high in the USA. Grasses that grow fast and produce large amounts of biomass are prime candidates for production of biofuel feedstocks. Annual and perennial grasses, particularly native grasses, are useful feedstocks for producing biofuel. Perennial grasses are advantageous over other biofuel feedstocks (e.g, crop residues) because they require low energy input for producing large quantities of biomass (Tilman et al., 2006).

Biofuel produced from grasses can be C-negative because they take up more C from the atmosphere through photosynthesis than released through the production cycle (e.g., CO_2, CH_4). Net gains in ecosystem C are due to that sequestered in soil as stable C, thereby increasing the soil organic pools. A diverse mixture of grasses can sequester 4.4 Mg ha^{-1} yr^{-1} of CO_2 in soil and roots which is higher than the 0.32 Mg ha^{-1} yr^{-1} of net fossil CO_2 released from biofuel production (Tilman et al., 2006). Well-structured and diverse grass species can produce usable energy, reduce soil erosion, and mitigate global climate change. Growing grasses are energy-negative because they require less input (e.g., chemicals, tractor fuels) than high-input biofuel crops (e.g., corn, soybean). Growing perennial grasses as biofuel feedstocks requires less agrichemicals than traditional crops, which would reduce water pollution. Perennial grasses are also beneficial to wildlife (flora and fauna) diversity. On a sandy soil in the USA, production of biofuel feedstocks increased with increase in the number of grass species grown and was 84% for 2 grass species, 100% for 4, 157% for 8, and 238% for 16 (Tilman et al., 2006).

Unlike food crops, another benefit of establishing diverse perennial grasses is that they can grow on marginal and degraded soils and be also used as biofuel feedstock. Growing a native perennial grass in degraded and marginal agricultural lands for biofuel feedstocks can save the land taken out of food crop production and thus release pressure on land resources. Moreover, growing native grasses is important to improving natural fertility of soils, restoring degraded soils, reducing non-point source pollution, improving wildlife habitat, and enhancing biodiversity. Native plant species are more resilient and can grow in harsh environmental conditions than most grain row crops.

Switchgrass, miscanthus, eastern gamagrass, big bluestem, and Indian grass are among the top biofuel feedstocks. In addition to these native warm-season grasses, a number of perennial cool-season grasses such as smooth bromegrass, Kentucky bluegrass, tall fescue, timothy, and birdfoot trefoil can be used as biofuel feedstocks (Florine et al., 2006). Cool-season grass pastures can be a viable alternative in ecosystems where production of warm-season grasses is not enough to meet the demands. Cool-season grasses can be harvested in late spring or early summer, whereas warm-season grass are harvested in late summer or early fall, thereby supplying biofuel feedstocks across different seasons. Other potential stiff-stemmed, annual or perennial herbaceous species for biofuel feedstock include New England aster, kinghead ambrosia, evening primrose, horseweed, cockleburr, field thistle, dames rocket, goldenrod, and wild sunflower (Kamm, 2004).

While excessive removal of grasses may adversely affect soil and environmental quality, controlled harvesting of grasses maintains wildlife habitat and even favors proliferation of birds and other wildlife animals that prefer short and moderate tall grasses. While much is known about the quality and quantity of grasses for forage production, little is known about the low input and high diversity grass species for producing biofuel feedstocks. More research data on the potential of perennial grasses as biofuel feedstocks based on the physiology of each grass species across different ecosystems and climate conditions must be pursued.

13.10 Methods of Grazing

- There are a number of methods of grazing based on the specific characteristics of each pastureland, ecosystem, and climate regime. No definite threshold stocking level exists for all ecosystems or even for the same ecosystem due to prevailing interactions among season, animal species, and vegetation growth.

Pastureland management falls primarily within two grazing techniques: *continuous* and *controlled stocking*.

- ***Continuous stocking*** refers to the unlimited access by animals to pastureland during a specified or unlimited period of time. This technique is simple and inexpensive, but it can adversely impact pastureland and soil productivity.
- ***Controlled stocking***, sometimes referred to as *controlled grazing*, unlike continuous stocking is a method that controls what and when animals graze on a specific piece of pastureland for optimizing pastureland production and sustainability. This technique matches up the number of animals with grass condition or growth by balancing animal requirements with grass and supplements. Remaining ungrazed pasture is harvested as hay or silage. Pasturelands under controlled stocking can use one or more grazing techniques within the same grazing system.

Some of the specific techniques of controlled stocking include the following (Johnson, 2003; Troeh et al., 2004):

1. ***Alternate stocking***. It refers to the repeated and successive grazing in two enclosed fields or paddocks.
2. ***Deferred grazing***. It consists of delaying the grazing period within a specific unit of pastureland in order to harvest seeds or allow grass species to reach their full potential of growth. This system is important because it allows grazing during times when grass is scarce. It can also be rotated among the different fields within the pastureland.
3. ***Intermittent grazing***. It is a system in which grazing is allowed at non-systematic intervals. It consists of removing stock for short periods of time (days or weeks) to reduce soil deterioration and overgrazing.
4. ***Mixed grazing***. In this system one or two more animal species are allowed to graze on the same unit of pastureland. Cattle, sheep, or mixed cattle + sheep are examples of this system. Animals are normally selective and differ in their consumption habits and grass species preference.
5. ***Ration or strip grazing***. In this system, animals are confined to a definite piece of pastureland to obtain their daily fodder allowance.
6. ***Rotational stocking***. This technique provides the greatest benefit because animals are shifted from one paddock or another during the grazing period following a systematic schedule.
7. ***Seasonal grazing***. This system allows grazing on the same piece land only during specific seasons of the year.
8. ***Sequence grazing***. It refers to the successive grazing of one or two more pieces of land with different types of grass species or forages. This system is beneficial because grass species differ in their composition, quality, and age, providing diverse benefits to animals.

13.11 Management of Grazing Lands

The goal of managing grazing lands is to achieve a desired outcome by managing all the major components such as soil attributes, plant species, and animals. The ultimate objective is to enhance grazing lands productivity, which is measured in terms of forage quality and quantity per unit of land. This obviously depends on grazing pressure that is number of animals per unit of land. Rotational or controlled grazing is a first choice to enhance a sustained production. Rangeland management involves the protection, improvement, rehabilitation of rangeland resources in order to obtain an optimum production while conserving soil and water, and improving biological diversity. Managing degraded rangelands requires an intensive planning and adoption of practices. Restoration of grazing lands seldom rests on a single practice.

The factors affecting management of pasturelands and rangelands include:

- Type of animal and stocking rate
- Grazing systems
- Grassland/rangeland size and production potential

- Pasture/forage quality and quantity
- Climate (e.g., temperature, rainfall distribution)
- Cultural-socio-economic characteristics
- Incentives and conservation programs
- Soil and landscape characteristics
- Mechanization/ modernization
- Grass species and distribution

13.11.1 Benefits of Grazing

Grazing of rangelands is one of oldest practices in the world and it does not necessarily reduce the potential of plant species. Indeed, moderate defoliation increases net production of the rangeland species by strengthening the stems and improving plant resilience to perturbations. Thus, prudent grazing is beneficial to maintaining an active ecosystem service and animal production. Rotational grazing is particularly effective at enhancing forage production and improving water quality. Amount and time of grass removal, plant physiological characteristics, and climate are critical factors that determine stress. In some ecosystems, grazing increases diversity of plants and reduces invasion by other species (Altesor et al., 2006). Animals constitute an integral component of the ecosystem because they recycle organic materials along with N and P compounds through fecal deposits. Controlled stocking is, thus, essential to reducing soil erosion, enhancing soil biological activity, and promoting C dynamics. The hoof action allows the spreading of seeds to maintain a diverse and mixed proportion of plant species. Finally, grazing is the source for all animal derived products essential to human consumption.

While overgrazing adversely affects soil properties, deposition of dung materials actually improves soil physical, chemical, and biological properties in localized spots in the field. Animal wastes concentrate as much as one billion organisms per gram of feces, and enhance soil microbial processes, soil aggregation, and nutrient recycling. Excessive input of animal wastes is a source of water pollution in modern agriculture, but proper management can reduce off-site movement of wastes while improving soil properties and ecosystem quality. Soil bulk density and water infiltration rates under patches of cattle dung are higher than those in soils without dung (Herrick and Lal, 1995). Fecal deposition by cattle also increases volume of macropores and drainable porosity. Amount and distribution of fecal materials and presence of microorganisms in the feces improve soil structural and hydrological properties.

13.11.2 Fire as a Management Tool

Use of controlled and well-timed fire is an important strategy to manage pasturelands except in arid ecosystems. The effectiveness of prescribed burning is a function of intensity and frequency of the fire. Burning vegetation stimulates regrowth, removes invasive species, and often increases grass production. Grasses growing on burned lands are more palatable, thereby attract more animals. Timely burning also reduces weed incidence and disease infestations. While untimely burning increases risks of soil erosion because it leaves bare areas, seasonal burning at the right time increases vegetation regrowth and quality and distribution of grass. Plant height should be small (<3 cm) enough at the time of burning to regrow rapidly. Seasonal burning of grasses is done in mid spring when plants are just greening up and when soils are moist enough to permit rapid regrowth. In midwestern regions of the USA, burning is done between March and April.

13.11.3 Resilience and Recovery of Grazed Lands

Plant species vary greatly in their resilience, tolerance, and competitivity under grazing pressures. Identification, knowledge, and development of phenotypic plant species tolerant to grazing for each rangeland ecosystem are important strategies to maintain an equilibrium between plant species and grazing. Concentration of carbohydrate and location and number of meristems are primary indicators of high buffering capacity of plants to grazing (Caldwell and Hodgkinson, 1986). Grazed plants must have morphological and physiological characteristics to compete with neighboring plants in the community. Some plant species survive under drought and large fluctuations in soil temperature. Stress attenuation also involves the ability of degraded rangelands to allow rapid growth from seeds and resistance of plants to adverse environmental conditions.

Avoidance and tolerance. Two main characteristics that enable a plant to resist/survive following perturbations are *avoidance* and *tolerance* (Briske, 1996). *Avoidance* refers to the ability of a plant to reduce defoliation by producing biochemical compounds in interaction with its morphological characteristics, while *tolerance* refers to the ability of a plant to regrow after grazing due to its specific physiological characteristics. Removal of stress (e.g., overgrazing) leads to a gradual recovery in forage production, species diversity, and plant population.

Resilience vs. ecological zones. Pasturelands in sub-humid, humid, and temperate regions are more resilient than those in arid and semiarid regions. Limited water, reduced soil development, steep landscapes, limited grass biodiversity, and extreme weather conditions in arid regions are among the factors that contribute to greater degradation and non-equilibrium conditions of pasturelands. Pastureland ecosystems are highly sensitive to harsh and variable climate conditions (e.g., drought) and do not always reach a steady state equilibrium in arid regions. Two dynamic components of pasturelands are *biotic* (vegetative cover) and *abiotic* (soil) entities (Perevolotsky and Seligman, 1998). Both components are sensitive to mismanagement and can be degraded rapidly. The difference is that the biotic component is highly resilient and recovers as fast as it is overgrazed or burned unlike the abiotic component, which requires a longer span of time (decades or centuries) to recover following degradation.

Rate of recovery. Removal of stress or overgrazing allows a rapid recovery. The amount of time needed for a system to recover is site specific. Some degraded pasturelands can recover within one year after exclusion but others require longer (>2 yr) periods of time. On a clay loam soil high in soil organic content in New Zealand, excessively sheep-grazed pastures produced higher sediment and nutrient concentrations in runoff than ungrazed pastures, but differences at 6 weeks following cessation of grazing were not significant, indicating a rapid recovery after heavy grazing was eliminated (Elliott and Carlson, 2004). Intensity of grazing, vegetation and soil type, topography, and climate influence the rate of recovery.

Threshold levels of recovery. There is a threshold level of resilience, beyond which a grazed land can not recover to its original state. Some systems do not recover to their antecedent its initial condition because of hysteresis resulting from reduced plant available water and nutrient levels.

Hysteresis. The hysteresis is "the degree to which the path of restoration is an exact reversal to path of degradation" (Westman, 1978). The pattern to which degraded pasturelands recover tends to be commonly lower than that of degradation unless the degree of degradation prior to restoration was only slight.

13.11.4 Conversion of Pastureland to Croplands

Conversion of pasturelands into croplands is a major anthropogenic cause of desertification of terrestrial ecosystems. This contemporary phenomenon has the most adverse impact on fragile and mountainous pastureland ecosystems. Mismanagement of converted lands has raised broad concerns of global soil degradation. Continued expansion of agriculture and the attendant reduction in pasturelands has increased risks of water and wind erosion, reduced soil fertility, and degraded environmental quality. Plowing former pasturelands destroys natural soil structure by breaking biologically bound aggregates and accelerating decomposition of soil organic matter (Table 13.2). It changes the stability and distribution of soil aggregates. Cultivation reduces formation of stable macroaggregates, increases soil erodibility, and reduces biological activities.

The organic matter content and the soil particle fine fraction (<0.1 mm) decrease over time in the surface layers with cultivation due to wind and water erosion.

Table 13.2 Changes in soil properties following conversion of pasturelands to croplands

Soil property	Pastureland	Cropland
Water content at -0.3 bar ($mm^3 mm^{-3}$)	0.3a	0.2b
Bulk density ($Mg m^{-3}$)	1.2a	1.4b
Cone index (MPa)	0.4a	0.7b
Water stable aggregates (%)	85a	35b
Organic C ($Mg ha^{-1}$)	30a	20b

Source: Blanco-Canqui et al. (2005) and Shukla et al. (2003). Means followed by the same letter within the same row are not significantly different.

Reduction in soil organic matter content further accelerates aggregate breakdown because macroagregation is positively correlated with organic matter content. Cultivated soils, especially those on steep terrain, are more erodible than pasturelands because of the lack of grass cover and interwoven root systems essential to stabilizing fragile and shallow soils. Reducing cultivation and adopting no-till systems are useful strategies to reduce degradation of former pasturelands. Soil aggregation and quantity of roots follow the order of pasturelands>no-till>plow tillage.

13.11.5 Conversion of Croplands to Permanent Vegetation

Conversion of croplands to pasturelands or native perennial grasses decreases runoff and soil erosion rates and restores soil properties with time to a degree similar to those under natural conditions. Soil type and resilience, length of time after conversion, and plant species determine the rate at which the soil recovers. High biomass producing and deep-rooted species develop biopores, improve aggregate stability, enhance water movement in the soil, and increase organic matter. Conservation programs, such as the CRP in the USA, require the conversion of degraded agricultural lands into permanent grasslands or native prairie grasses to stabilize erodible soils and improve soil quality. Changes in soil properties with land conversion to CRP are not always rapid and linear. Conversion of erodible plowed croplands to grasslands and no-till can reduce sediment losses by 40 and 80% because of higher amounts of undisturbed above- and below-ground biomass (Yuan et al., 2006).

13.11.6 Rotational Stocking

This grazing method is advantageous over continuous rotation because it optimizes forage production. Rotational grazing has greater vegetative cover and less weed invasions than continuous grazing (Fig. 13.6). Intensive rotational stocking is a common technique adopted by livestock producers to reduce costs of feeding, machinery, and labor. This intensive grazing management can, however, damage the grass cover. Intensive short-duration grazing systems can indeed have more adverse effects on soil properties and forage production than moderate continuous grazing systems. Parameters such as height of plant before and after grazing are a simple approach to determine grazing regimes based on plant morphology and animal stocking (Carlassare and Karsten, 2002). Recommendations on the height of grazing depend on the height and morphology of plant species. For example, bunch-type grasses (e.g., orchardgrass, switchgrass, big bluestem) are taller than sod-forming grasses (e.g., tall fescue, Kentucky bluegrass). Animal intake increases with increase in plant height because of increased availability and accessibility. Current recommendations of grazing heights for tall grasses vary between 18 and 30 cm before grazing and between 5 and 7.5 cm after grazing, depending on local soil and climate conditions (Hall, 1998).

Fig. 13.6 Rotating sheep from one cell to another is a strategy to manage pasturelands (Courtesy USDA-NRCS)

13.11.7 Restoration of Degraded Grazed Lands

Restoration denotes the accounting of environmental, economic, and social conditions. Parameters of both macroscale (e.g., climate) and microscale (e.g., soil type, livestock number, and site-specific management) affect restoration and management of degraded lands. Knowledge of livestock stocking rates and nutritional requirements must match the adaptability and diversity of plant species (Table 13.3). Landscape characteristics and magnitude of soil erosion and overgrazing determine the resilience of vegetation. Temporal or permanent closure of degraded areas is recommended to allow recovery. Management of pastureland resources is intrinsically related to livestock, agriculture, and urban development. The relationship among these resource components must be understood to predict and design proper pastureland development options. Control of soil erosion from pasturelands is a function of agronomic measures, and a major obstacle for proper management is the lack of guidelines for sustainable management for diverse eco-regions.

Strategies of manage pasturelands include fertilization, weed control, reseeding, restoration, and controlled grazing.

Some of the measures to restore degraded grazed lands include:

1. Implementation of planned grazing management (e.g., herding),
2. Restoration of degraded pasturelands with native and perennial grass species rather than with short growing grass species as native species are more environmentally desirable,

Table 13.3 Biotic and abiotic factors influencing grass reestablishment in degraded ecosystems

Soil and climate	Management	Livestock	Vegetation
• Topography • Plant available water capacity • Compaction • Salinization • Drought and flooding risks	• Harvesting • Mechanization • Invasive species • Fertilization • Drainage systems • Fires • Overgrazing	• Stocking rates • Forage preferences • Trampling potential	• Type • Diversity • Availability • Resilience • Adaptability

3. Establishment of conservation buffers and construction of streambanks and ponds,
4. Regulation of the stocking rate and distribution of livestock. In some systems, traditional grazing with mobile herds is more easily manageable,
5. Establishment of rotational grazing and construction of fences and shifting the animals from paddock to paddock and backfencing fields to account for the spatial heterogeneity of fields and differences in grass production from year to year,
6. Construction of stream paths or livestock crossings and establish drainage systems to reduce pugging and compaction. The greater the animal traffic, the more is the soil deterioration and grass damage,
7. Improvement of degraded pasturelands by seeding select non-invasive grass and legume species,
8. Use of supplements with hay or silage as an alternative to grazing in wet soils,
9. Installation of backfence paddocks to prevent backtracking over the grazed areas which may cause further pugging and reductions in plant growth,
10. Increase of the length of rotational grazing to allow the regrowth of grass and increase the grazing area to reduce stocking density,
11. Allowing grazing when grass is sufficiently mature or tall (10–15 cm). Animals do not walk over large distances when grasses are tall, reducing trampling and pugging. Taller plants also recover faster from grazing unless they are buried by traffic, which can rot the grasses,
12. Improvement of paths of access to drinking water locations and isolate sensitive or eroded areas,
13. Application of N when soil is not waterlogged to reduce N losses while maintaining proper levels of P and K. The N applied early (autumn, early winter) will aid the build up of a feed wedge for winter, and
14. Using controlled application of animal manure and compliance with the conservation or pastureland management programs.

13.12 Modeling of Grazing Land Management

Pastureland assessment and management using traditional approaches are often difficult. Models permit the integration of all complex and interrelated components of the ecosystem. Use of models is a new paradigm for an optimum management of pasturelands. Sophisticated modeling approaches and remote sensing are useful tools to characterize and manage the various scenarios of pastureland and rangeland management. Use of GIS to integrate data on plant species, soil type and slope, extent and age of rangelands allows the assessment of rangeland productivity and soil degradation problems. Modeling approaches also permit anticipation of future trends in management. High resolution images with differing spectral bands and radiation features are useful to estimate plant height, canopy structure, bare soil patches, soil roughness, and soil erodibility (Hunt et al., 2003). Combination of remote sensing with models using GIS approach is important to assess and monitor rangeland characterization.

Some of the computer-based models of pastureland, forage, and livestock production include RAPS (Resource Assessment for Pastoral Farming Systems), GRAZE, and GRAZFEED, whereas models of pasturelands in relation to soil erosion include SWAT, WEPP, GLEAMS, and RUSLE (FAO, 2000). Hydrologic models in combination with GIS and satellite images have been used to assess impacts of converting pasturelands to agricultural and urban areas on runoff rates and flash floods and water quality in northern Mexico and southwestern USA (Miller et al., 2002). Soil erosion models are important decision tools because they predict the likelihood of runoff and soil loss rates by incorporating parameters of surface vegetative cover. For example, the C-factor in RUSLE simulates the relative effects of decreasing grass or forage cover by grazing and burning on soil erosion. The C-factor integrates information on canopy grass cover, root biomass, soil surface cover or litter, soil roughness, soil water content, and previous land use (Refer to Chapter 4). Monitoring and estimating soil erosion are critical because plant growth is directly influenced by the magnitude of soil erosion. Models are essential to evaluating and gauging the potential of rangelands for a sustained production. Soil erosion rates are sensitive indicators of upland management and their increases indicate the vulnerability of the systems and signal the need for modifying current management systems.

Models allow the assessment of:

1. Potential of grazing lands to produce forage and support livestock requirements over time,
2. Performance of past, present, and future management systems,
3. Performance of management systems, new and improved forage production systems, introduced grass species, different grazing systems, and changes in production systems, and
4. Impacts of grazing on soil and water quality, wildlife habitat, biodiversity, and overall environmental quality.

Summary

Pasturelands, grasslands, meadows, and rangelands are important ecosystems for soil and water conservation and livestock production. Pasturelands comprise single or native grass species known as “grasslands”, which cover about 40% of the earth’s terrestrial ecosystem. Pasturelands are crucial to improving environmental quality by reducing water and wind erosion, sinking pollutants, sequestering C, and buffering the whole ecosystem. Rangelands are more complex ecosystems than pasturelands consisting of native grasses, grass-like vegetation, forbs, shrubs, and woods. Rangelands are often not cultivated and animal stockings rates are lower than those in pasturelands. Pasture and rangelands are affected by intensive cultivation, intensive grazing, fires, road constructions, and introduction of invasive plant species.

The capacity of these lands to provide ecosystems is declining due to rapid degradation and fragmentation. Anthropogenic activities have increased soil erosion rates, off-site transport of pollutants, and decline in diversity of flora and fauna, whose magnitude depend on management (e.g., commercial, traditional) and climate. Intensive grazing causes soil erosion, reduces soil organic matter content, and deteriorates soil physical, chemical, and biological properties. Well-managed grasslands reduce runoff and soil erosion by protecting the soil surface and stabilizing soil matrix through physical, biological, and chemical bonding mechanisms.

There are two main methods of grazing continuous and controlled stocking. Controlled stocking include alternate stocking, deferred grazing, intermittent grazing, mixed grazing, strip grazing, rotational stocking, seasonal grazing, and sequence grazing. The rate of recovery of degraded pasturelands depends on grazing intensity, topography, management, type of livestock, and vegetation. Perennial warm- and cool-season grasses can be potential biomass feedstocks for biofuel production in the near future. Growing grasses in marginal and degraded soils as biofuel feedstocks may be more economical than using agricultural crops. Models are important tools to characterizing and managing the various scenarios of pastureland and rangeland management because they allow the assessment of performance of past, present, and future management systems.

Study Questions

1. Determine if vegetation along a 6% slope pastureland channel under tall fescue fails under concentrated flow. The runoff flow depth, which varies during the rainstorm event, is 0.1, 0.25, 0.5, and 1 m. The rigidity constant (MEI) is 30 while the channel hydraulic radius is 3 m.
2. Calculate the friction factor the grass in Prob. 1 if the height of ungrazed is 0.5 m and 0.1 m when grazed.
3. Compute the runoff flow velocity if the Manning's coefficient of roughness for the vegetation is 0.06.
4. Discuss difference between pasturelands and rangelands in terms of magnitude of soil erosion.
5. Discuss differences among set-stocking, continuous stocking, and rotational stocking.
6. Discuss how modeling can contribute to better management of pasturelands and rangelands.
7. Explain the potential of grasslands for providing feedstocks for producing biofuel.

8. How does the Manning's roughness coefficient change with overgrazing?
9. Describe the strategies for managing degraded pasturelands.
10. Explain the mechanisms by which the grass roots reduce runoff and soil erosion.

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Chapter 14
Nutrient Erosion and Hypoxia of Aquatic Ecosystems

Freshwater is a finite resource. Only about 2.5% of the total volume of the global water reserve is freshwater, and the remainder (97.5%) is the oceanic brackish water (IUCNN, 2003). The usable freshwater is less than 1% of the total freshwater. Agriculture accounts for about 85% of the global demand for the freshwater resources (Foley et al., 2005). The major sources of freshwater include lakes, streams, rain, snow, and the soil water reserve. Freshwater demands have increased during the 20th century especially for agriculture (e.g., irrigation), human consumption, and industrial uses, thereby affecting competition for the scarce water resources. Because of the excessive use, some rivers, especially in semiarid regions, have lower flow rates than before and are prone to being ephemeral. Excessive consumption and misuse of water have also reduced the groundwater level. Conservation and proper use of freshwater resources are essential to attain sustainable use.

14.1 Water Quality

Coincident with the decline in surface and ground water resources, there is a growing concern about the eutrophication and contamination of water resources. Anthropogenic activities are mainly responsible for the decline in water quality. Intensive cropping, livestock raising, increased use of pesticides and fertilizers, and lack of effective erosion control practices have contributed to pollution of rivers, lakes, and subsurface waters through transport of pollutants in surface runoff and seepage water. Significant quantities of agricultural nutrients and pesticides, in excess of the maximum permissible levels, are transported into the drinking water supplies in many rural communities (USEPA, 2008a). Modern agriculture has led to expansion of the farm size, intensification of tillage, and row cropping to realize high production levels. Intensive farming and excessive plowing have altered the soil surface conditions and increased rates of runoff and sediment losses. Water pollution is particularly severe in regions with excessive indiscriminate use of agricultural chemicals. Water runoff transports sediment and nutrients such as N and P into aquatic ecosystems (e.g., streams, rivers, and lakes). Excess of NO_3^- in drinking water can trigger major health problems such as methemoglobinemia that causes the death of infants <6 months old (Mitsch and Day, 2006).While adoption of modern technologies have revolutionized agriculture and increased food production, these have also led to a severe steady decline in water quality. Thus, restoring water resources and enhancing water quality are high priorities. Water pollution is also a major problem in developing regions with limited access to water treatments plants (Fig. 14.1).

Fig. 14.1 Water contamination in developing countries (Photo by H. Blanco)

14.2 Eutrophication

Eutrophication refers to the increase in concentration of plant nutrients in water bodies which results in excessive growth of certain organisms, depletion of dissolved O_2, and elimination of aerobic organisms (USEPA, 2008a). It can occur naturally through the gradual aging of the water bodies or rapidly through human-induced sources of nutrient pollution. The process of eutrophication by natural means is slow and often reversible, but anthropogenic activities have accelerated the process through N and P input and diminished the buffering capacity of the ecosystems. The higher concentrations of dissolved nutrients in water (e.g., NO_3-N, PO_4-P) promote rapid growth of algae. The hyper-growth of algae chokes and suffocates other plants. Eventual death of algae uses up most of the available O_2 in water, reducing O_2 levels to minimum, and jeopardizing the aquatic life. Both plant nutrients and sediment contribute to eutrophication of waters in lakes, ponds, and bays. Severe eutrophication leads to water impairment for drinking, industry, agriculture, biodiversity, and recreation. Unlike other types of pollution, eutrophication primarily refers to pollution of water by excessive concentration of plants nutrients.

14.3 Non-point Source Pollution and Runoff

Pollution refers to “the presence or introduction of a pollutant into the environment.” (SSSA, 2008). Common pollutants include sediment, nutrients, pesticides, animal wastes, and household or industrial products (e.g., paints, motor oil, antifreeze). There are two main sources of pollution: point source and non-point source pollution. The point source refers to the water pollution that originates from known or definite sources such as discharge pipes of wastewater from factories, and industrial and municipal sewage facilities. In contrast, non-point source pollution originates from diffuse source and can not be traced to a single or identifiable source (USEPA, 2008a). The non-point source pollution is a major cause of water contamination, in which pollutants enter streams, lakes, and other water sources via runoff (e.g., sediment-bound, dissolved nutrients) and atmospheric deposition (e.g., airborne pollutants). The pollutants travel through storm drains, ditches, and culverts, and are deposited in rivers and streams. Airborne pollutants derived from gas emissions of automobiles, industrial plants, and wastewater treatment plants are returned as acid rain to soil and water. Pollutants can also be carried by wind and deposited in waters. Pathogens carried in runoff from untreated sewage, septic systems, and livestock farms can also pollute natural waters.

14.4 Factors Affecting Transport of Pollutants

The build-up, transport and delivery rate of non-point source pollutants are complex processes controlled by a range of factors including types of pollutants, land use, soil characteristics of the source area, soil management, and climate (Table 14.1). Bare soil surface conditions, high rainfall, steep topography, and inadequate soil conservation practices are contributing factors to higher runoff. Direct raindrop impacts on bare soil detach soil particles and initiate transport of detached particles and pollutants in runoff. Management practices that leave little or no residues on

Table 14.1 Determinants of non-point source pollution

Soil mismanagement	Soil characteristics	Climate	Use of fertilizers and pesticides
• Monocropping • Crop residue removal • Low vegetative cover • Excessive tillage • Row cropping	• Low organic matter content • Steep slopes • Poor internal drainage • High silt content • Low permeability • Low activity clays	• Intense rains • High runoff potential • High wind velocity	• High rates of fertilizer application • Poor timing of chemical application • Poor selection of herbicides • Surface application of chemicals • High solubility of chemicals

the soil surface increase runoff and sediment losses. Crop residues or plants absorb raindrops, water infiltration, and runoff.

14.5 Pollutant Sources

- ***Agriculture.*** Sediment, N, and P loss in runoff from agricultural lands is the origin of much of non-point source pollution around the world (Table 14.2). In the USA, about 60% of sediment, 82% of N, and 84% of P in surface waters are from non-point sources comprising of agricultural lands (USEPA, 2008b). Forty percent of U.S. rivers and lakes have excessive amounts of sediment, N, and P, and are serious health threat to human, wildlife, livestock and aquatic flora and fauna (U.S. Geological Survey, 1999). Large amounts of sediment and sediment-borne and dissolved chemicals are washed off cultivated fields in runoff water (Fig. 14.2). Indiscriminate disposal of animal wastes is another major source of water pollution.
- ***Urbanization.*** Fertilizers and pesticides from gardens and lawns, and pollutants such as automobile oils, battery acids, cleaners, and other household/industrial chemicals that end up in runoff water contribute to water pollution. Runoff from urban areas (e.g., industrial yards, parks, sport fields, railways, driveways, sidewalks, parking lots) collects and delivers a wide range of organic and inorganic pollutants to downstream water sources. Municipal waste treatment plants and septic systems can also pollute water if improperly designed, poorly maintained or improperly managed. Storm sewer and wastewaters discharges can be transported in surface runoff or seep into the groundwater and contaminate wells or subsurface waters. Construction sites are also a source of sediment and organic materials (e.g., oils, paints). Poor timing and stabilization of disturbed areas exacerbate sediment load.
- ***Acid precipitation.*** It refers to the return of chemical substances in the form of acids via precipitation or wind. Smoke and gases emitted mainly by industrial plants and automobiles that use fossil fuels react with rain water, form both

Table 14.2 Some common point and non-point source of pollutants

Agricultural lands	Non-agricultural lands	Urban ecosystems
Croplands: • Sediment • Inorganic fertilizers • Herbicides • Insecticides Livestock Manure: • Cattle, hog, and sheep • Poultry farms	Natural Ecosystems: • Pasturelands • Rangelands • Abandoned lands • Forest lands Mined Soils: • Sediment • Acid drainage	Municipal/Construction Sites: • Sewer discharges • Oil and grease • Toxic chemicals • Industrial effluents Residential Areas: • Gardens • Lawns • Recreational lands

Fig. 14.2 A waterway polluted with sediment from an adjacent cropland (Courtesy USDA-NRCS). Pollutants (e.g., pesticides, herbicides, N, P) are transported with sediment particles in runoff. Sediment is the product of runoff and soil erosion

strong and weak acids, and are returned to soil and water systems as pollutants. Greenhouse gases such as CO_2, sulfur dioxide (SO_2) and NOx are the main causes of acid precipitation. In the U.S., about 65% of all SO_2 and 25% of all NOx are produced by electric power plants that use coal (USEPA, 2008c). These gases in reaction with water form carbonic, sulfuric, and nitric acids. The CO_2 emitted from the decomposition of organic material, forms carbonic acid, a weak acid, and is a source of natural acidity in rainwater. Because carbonic acid is a weak acid, the current large emissions of CO_2 from terrestrial systems may particularly be more of a concern in relation to the projected global warming than in regards to water pollution. The pH of acidic rain with sulfuric and nitric acids can be as low as 4. Acids deposited on the soil surface are carried to streams and lakes by runoff. Acidic gases and fine particles carried by prevailing winds are often deposited on vegetation and urban lands. Such depositions are washed off by rain and runoff. Acid rain is among major global pollutants, which often goes undetected but has strong impact on ecosystem quality. The incidence of acid rains, particularly in highly industrialized regions, has increased since 1970s (van Breemen and Wright, 2004). The extent and severity of damage depend on the amount and type of acids and the buffering capacity of the soil and ability of plants and animals to tolerate acidity.

- ***Mining.*** Drastic land disturbance, poor construction and operation, and improper reclamation of minesoils also cause pollution of natural waters. Disturbed minelands are a major source of large amounts of sediment in runoff, especially when the disturbed land is not immediately reclaimed. Also, coal mines generate strong acid effluents that are washed into streams, lakes, and ponds or seeped into groundwater. Heavy metals from abandoned mines (e.g., Hg, As, Cu, Zn, Fe) are major components of acid drainage in minesoils (Sheoran and Sheoran, 2006).

- ***Other sources.*** Sediment transport from some uncultivated lands is also important source of pollution. Logging operations and construction of access roads on forest lands, for example, exacerbate soil erosion and increase sediment transport to streams and lakes. Any perturbation of the soil accentuates dissolved and suspended loads and increases risks of water pollution. Activities including stream channel dredging and channelization can generate sediment and affect the aquatic ecosystems. Construction of roads, bridges, dams, and diversion channels removes the natural vegetation and adds sediment to waters, especially if improperly designed and poorly managed. Trees and grasses along streams and lakes stabilize banks, improve wildlife habitat, and reduce streambank erosion.

14.6 Common Pollutants

Sediment transport is a selective process. Fine particles (e.g., clay, fine silt) are transported more easily and over large distances than coarse particles. Besides, fine and less dense particles remain in suspension for a long time and are transported over long distances in runoff compared to coarse particles. According to the Stoke's law, large particles settle faster than small particles. Fine particles, however, have much higher specific surface area than coarse particles and can absorb more ionic forms of nutrients and pesticides. This means that greater amounts of chemicals are transported in runoff through fine particles per unit mass of soil than through coarse particles. Particulate organic matter is also transported more readily than mineral particles because of its low density. High surface area and charge density of particulate organic matter increase absorption and transport of agrichemicals because of its high adsorption capacity. In general, sediments contain higher concentrations of nutrients and pesticides than the original soil (USEPA, 2008b), because of selective removal and high enrichment ratio of pollutants.

14.6.1 Sediment

Sediment refers to transported and deposited particles or aggregates derived from rocks, soil, or biological material. It comprises of suspended materials including clay, silt, and other solids that are washed off arable lands, land-cleared sites, construction sites, strip-mined lands, logging sites, overgrazed lands, disturbed stream banks, and other anthropogenically disturbed sites. Sediment transports nutrients, toxic substances, and chemicals to rivers, lakes, and coastal waters. Because of high turbidity, sediment in water reduces visibility and decreases light penetration, adversely affecting the aquatic life (e.g., coral reefs). If there is no water or wind erosion, sediment delivery to streams and lakes would be practically negligible. Sediment transport associated with gravity or ice is much lower compared to water and wind erosion. Soil erosion from agricultural lands is the largest source of sediment through interrill, rill, and gully erosion. Interrill and rill erosion mainly remove topsoil, which is rich in agrichemicals, organic matter, and surface dwelling soil organisms. In contrast to interrill and rill erosion, gully erosion is more severe and removes sediment both from topsoil and subsoil from gullied agricultural and non-agricultural lands.

14.6.2 Nitrogen

While N is an essential macronutrient for crop production, its excessive use is a principal cause of water pollution. Until the mid 1950s, excess N was uncommon and most of it was provided by N-fixing organisms through biological nitrogen fixation. The introduction and widespread use of inorganic/synthetic N fertilizers have exacerbated the problem of eutrophication of water. Consumption of N fertilizer has increased steadily worldwide since 1960s although it has slightly decreased or stabilized in the developed countries during since 1990s (Fig. 14.3) (USDA-ERS, 2005; IFA, 2006). In the USA, corn production uses the largest amount of N fertilizers followed by wheat. Increases in N fertilizer use are correlated with increases in crop yields (Fig. 14.4) (USDA-ERS, 2005). The N fertilizer use increased from about 30×10^6 Mg in 1960 to 90×10^6 Mg in 2005 in the world and from 3×10^6 Mg in 1960 to 12×10^6 Mg in 2005 in the USA (Figs. 14.3 and 14.4). Concentrations of N in surface waters have also increased in accord with the increase in N fertilizer use. Not all the N fertilizer applied to the soil is absorbed by the plants during a growing season. Excess inorganic and organic N are carried in runoff to nearby water courses.

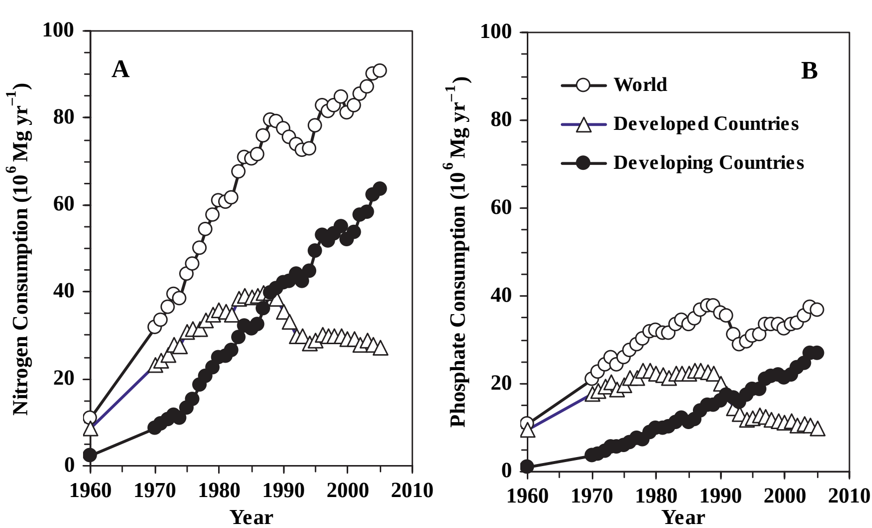


Fig. 14.3 Consumption of (**A**) nitrogen and (**B**) phosphate fertilizers in the world (After IFA, 2006)

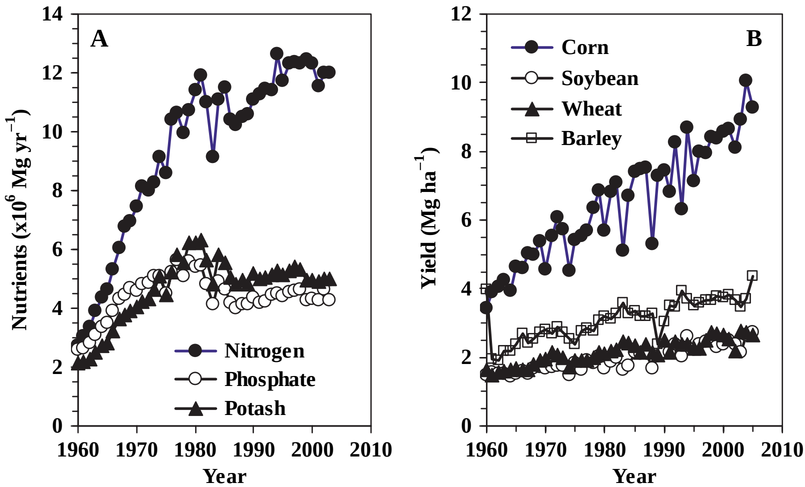


Fig. 14.4 (**A**) Consumption of fertilizers and (**B**) production of main crops in the USA (After USDA-ERS, 2005)

Low concentrations of ammonium (0.2 mg L^{-1}) impede most of the aquatic life and contribute to the development of polluted environments. The upper limit of nitrate-N in water used for human consumption is 10 mg L^{-1} (USEPA, 2008a). Organic N is transported with soil runoff sediment and organic matter while inorganic forms (e.g., ammonium, nitrates, nitrites) are transported in dissolved forms. Nitrates are highly mobile and can be leached out of the root zone whereas ammonium is often adsorbed by soil particles.

14.6.3 Phosphorus

While P is an essential nutrient for plant growth, application of excessive rates of P fertilizers and animal manure to agricultural lands has, however, increased the loadings of dissolved and particulate P, causing euthropication of surface waters (Fig. 14.5). Phosphorus levels, as low as 0.01 mg L^{-1} in water, can cause euthrophication and induce algal blooms. Repeated applications of fertilizers and manure to meet the N requirements have over-enriched the soil with P due to divergences between N:P ratio in manure (3:1) and that in crop requirement (8:1) (Grande et al., 2005). The use of P fertilizers is less than that of N (Fig. 14.3). It increased from about 1×10^6 Mg in 1960 to 27×10^6 Mg in 2005 in developing countries, and has tended to decrease in developed countries.

Because the ionic forms of P are strongly adsorbed on clay particles, transport of P by leaching is small and mainly occurs with sediments. The strong adsorption of P by clay has caused the enrichment of soil P levels. Additions of P-enriched manures have created surplus P, increasing P concentrations in runoff and aquatic ecosystems. Although most (>75%) of the P is transported as bound to mineral and

Fig. 14.5 A lake severely affected by algae blooms (Courtesy USDA-NRCS)

organic particles, a smaller yet significant fraction is transported as dissolved P in runoff. Thus, reductions of soil erosion may not significantly reduce transport of dissolved P. For example, no-till practices which dramatically reduce soil erosion and sediment-bound nutrients may still lose significant amounts of dissolved P in runoff water (Grande et al., 2005). Furthermore, clay particles in runoff are not filtered as easily as silt and sand particles by residues or grass hedges or strips. Indeed, P-adsorbed clay is easily transported off-site in runoff. Thus, transport of P in runoff bound to clay particles can be significant. Strong absorption of P on suspended clay is attributed to its high specific surface area and sorption capacity.

14.6.4 Animal Manure

Livestock manure is a valuable source of organic matter and macro- and micronutrients essential for plant growth. Manuring improves soil structure, aggregation, water infiltration, and thus reduces runoff and sediment losses. While manure is an excellent additive to improve soil, it can be a major pollutant of water and air. The rapid shift from numerous to fewer but larger livestock operations with high concentration of animals has raised concerns over use and disposal of the large quantity of manure generated. Runoff from manured fields usually contains higher concentrations of N and P than that from unmanured land. Surface application of manure to no-till soils may induce N losses by runoff and volatilization because manure is not incorporated into the soil (Eghball et al., 2000). Indeed, applications of dry manure at $>8\,\mathrm{Mg\,ha^{-1}}$ increase nutrient losses in no-till as compared to plowed soils (Mueller et al., 1984). Losses of manure-derived nutrients in runoff from no-till soils can be, however, reduced by improvements in soil properties by manuring.

Livestock production systems including hog, beef cattle, dairy cattle, sheep, and poultry farms generate large amounts of manure and wastewater and constitute a major source of impairment of drinking water supplies and pollution of aquatic ecosystems. Indiscriminate application of manure and improper management of wastewater cause over-enrichment of surface water and groundwater with nutrients (USEPA, 2008d). Livestock production techniques have changed significantly during since 1980s. Small- and medium-sized animal farms have gradually been replaced by large livestock operations. While the total number of livestock remains nearly the same, the density of animals in confined livestock operations has increased. Livestock concentration in high-production areas often exceeds 16 confined animals units per hectare for cattle operations (USEPA, 2008d).

The continued expansion of poultry industry also generates large amounts of by-products such as broiler litter. Thus, concerns over an environmentally sound way to dispose the broiler litter are also increasing. Broiler litter consists of manure and bedding material which must be periodically removed from poultry facilities. Because broiler litter is very rich in nutrients and organic matter, its by-product is often applied to crop and pasture lands and can be a source of water pollution. The water pollution with manure-derived nutrients is a serious concern in the USA and other industrialized countries.

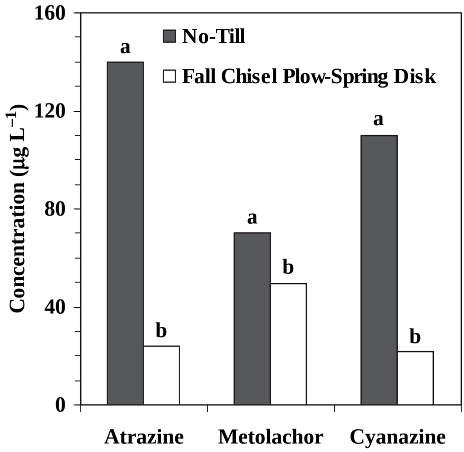
14.6.5 Pesticides

Use of pesticides (e.g., herbicides, insecticides, fungicides) is common in modern agriculture and has increased crop yields. Excess pesticides are, however, transported to water bodies by runoff and leaching. The same processes that affect soil transport are involved in the movement of pesticides. In the USA, as much as 95% of the streams and 60% of shallow wells reportedly have high concentrations of pesticides (Hamilton and Miller, 2002).

Herbicides are among the most commonly used pesticides. Herbicide concentration in runoff immediately after application can exceed the maximum permissible levels for drinking water. The standard levels for three major herbicides used in the U.S. Corn Belt region are $3\,\mu g\,L^{-1}$ for atrazine, $1\,\mu g\,L^{-1}$ for cyanazine, and $70\,\mu g\,L^{-1}$ for metolachlor (Mickelson et al., 2001). About 95% of the cultivated area in the U.S. Corn Belt region receives herbicide application. A significant portion of herbicides is usually lost in runoff occurring immediately after application. About 315 different types of herbicides are available in the U.S., and their use is more than those of fungicides and insecticides (WSSA, 2006). Most of these herbicides have been detected in surface and subsurface waters.

Choice of herbicides depends on the weed species, soil type, tillage method, cropping system, and climate. Conservation tillage such as no-till relies heavily on herbicides to control weeds. While conservation tillage significantly reduces soil

Fig. 14.6 Herbicide losses as affected by tillage management on a loam soil in the U.S. Corn Belt region (After Mickelson et al., 2001). Bars with different letters, within each herbicide type, indicate significant differences ($P < 0.05$)



erosion and runoff, herbicide transport may not always be reduced as compared to the plow tillage systems. In some soils, herbicide losses from no-till fields are higher than from plowed soils (Fig. 14.6).

There is a direct and positive relationship between runoff and herbicide transport in that lower water runoff result in correspondingly lower herbicide transport. Herbicide concentration in runoff can be computed as per Eq. (14.1) and (14.2) (Ghidey et al. 2005):

$$[C] = [C_0] \times \exp(-k \times t) \tag{14.1}$$

$$[C] = a \times \left(\frac{R}{Q}\right) \times [C_0] \times \exp(-k \times t) \tag{14.2}$$

where C is herbicide concentration (µg ha^{-1}), C_0 is initial herbicide concentration, R is herbicide application rate (µg ha^{-1}), Q is runoff volume (L ha^{-1}), a and k are coefficients, and t is time.

14.7 Pathways of Pollutant Transport

The main pathways for the transport of pollutants from the source areas include surface runoff, subsurface lateral flow, leaching, and volatilization (Fig. 14.7). Water and wind transport the pollutant to surface and ground waters. The amount of pollutants transported in surface runoff is often much higher than that through lateral flow, leaching, and volatilization. Rate and magnitude of pollutant transport from plots, watersheds, and fields depend on the site-specific conditions (e.g., soil characteristics, amount and type of pollutant, time and method of application, rainfall, wind). Soil characteristics (e.g., texture, clay mineralogy, permeability, organic matter content) affect the sorption of chemicals. Clay and organic matter adsorb

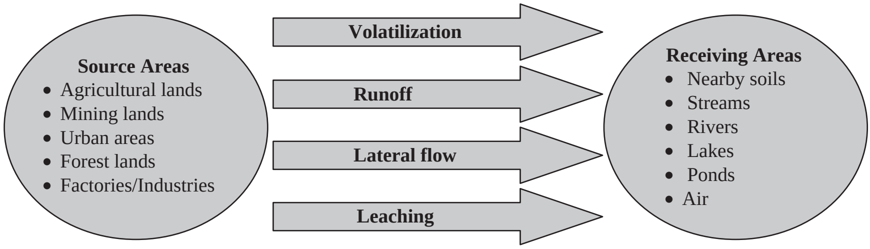


Fig. 14.7 Pathways of pollutants from source areas to nearby systems

large quantities of chemicals because of their high specific area. Depending on the solubility and formulation, adsorption varies with intrinsic characteristics of nutrient and pesticide. Strongly adsorbed chemicals by soil particles are not easily leached or transported in runoff water.

14.7.1 Water Runoff

Runoff water from rainfall or snowmelt is the principal mechanism of pollutant transport. It carries dissolved, suspended, and sediment-bound pollutants, and it deposits them into downstream water bodies. Off-site transport of pollutants via runoff is a major concern because large amounts of pollutants can be carried even by a single runoff event. Runoff water preferentially removes the pollutants that are at or near the soil surface. Thus, the total concentration of pollutants transported in each runoff declines over time. The concentration of nutrients and pesticides in the 10 mm of surface soil determines the amount of chemical transported in runoff (Gevao and Jones, 2002). Chemicals that are spread and left on the surface are transported in runoff more rapidly than those plowed under due to differences in adsorption rates. Adsorbed chemicals (e.g., P) are commonly transported with sediments. Runoff from urban areas also contributes to the total surface runoff for transporting pollutants. Urban runoff contains numerous pollutants both from impervious surfaces and vegetated areas. In layered soils (e.g., claypan), lateral flow can be important component of total runoff and pollutant transport.

14.7.2 Leaching

Leaching is another important pathway for transport of pollutants into natural waters. Dissolved nutrients and pesticides are transported downward by percolating water and by diffusion in gaseous phase. Soil structural, chemical, and biological properties, and rainfall amount interactively influence leaching of chemicals (Table 14.3). Pollutants preferentially leach through cracks, macropores, and root passageways. Pore connectivity, continuity, and size control pollutant movement within the soil. Earthworm burrows, for example, are main conduits for preferential

Table 14.3 Factors influencing pollutant leaching

Climate	Soil characteristics	Application of chemicals
• Rainfall amount and intensity • Frequency of rainfall • Wind velocity and temperature • Evaporative demand	• Texture and structure • Hydraulic conductivity • Infiltration rate • Drainage class • Profile characteristics • Clay mineralogy • Pore-size distribution • Macroporosity	• Amount of application • Formulation • Solubility • Half life or residence time • Rate of adsorption • Method of application • Timing of application

or by-pass flow of pollutants to deeper soil horizons. Preferential flow reduces soil matrix and pollutant interaction essential to chemical adsorption and degradation. Chemicals with higher sorption properties are less prone to leaching. By-pass flow accelerates the delivery of pollutants to subsurface water sources. While macropores are important to water recharge, they can be a problem in production systems with high chemical inputs (Shipitalo et al., 2000).

14.7.3 Volatilization

Volatilization of N from fertilizers and surface applied animal manure is a major cause of air pollution. Ammonia (NH_3) from manure and the breakdown of urea is emitted into the atmosphere, especially when soil$\times$N interaction is minimal. Emissions of NH_3 also contribute to eutrophication of surface waters and soil acidification through deposition. About 400,000 Mg yr^{-1} or 63% of N is volatilized from swine manured fields (USEPA, 2004). In the USA alone, about $80 million worth of fertilizer is lost annually because of NH_3 volatilization from manured fields. Spreading manure onto the soil surface exposes manure to the climatic elements and causes rapid emissions of NH_3. About 25% of the N in surface applied manure is emitted by volatilization as ammonium within 24 h after surface application (USEPA, 2004). The rate of volatilization is a function of soil surface water content, temperature, and pH. Volatilization of ammonia from manured fields is the highest under dry, warm and windy climates and from coarse textured soils. Gentle rains and cool temperatures following surface application of chemicals may minimize volatilization. The rate of NH_3 volatilization is increased with temperatures ($>10^{\circ}$C) and soil pH values (>7) (USEPA, 2004).

14.8 Hypoxia of Coastal Waters

Hypoxia refers to the depletion of dissolved O_2 in water, resulting from eutrophication. It occurs when the O_2 levels drop below 2 mg l^{-1}, a level that is too low to support the aquatic life. Development of hypoxic zones is a threat to ecosystem quality and economy of regions in proximity of the coastal zones. The problem of hypoxia affects many coastal areas around the world (e.g., USA, Europe). The O_2-depleted areas in coastal waters and bays have increased since the 1960s (Malakoff, 1998). About 150 hypoxic zones have been identified worldwide covering more than 70,000 km^2 (Boesch, 2002). Some of the coastal areas and bays affected by hypoxia include the Gulf of Mexico, Baltic Sea, northern Adriatic Sea, Gulf of Thailand, Yellow Sea, and Chesapeake Bay (Boesch, 2002). Large hypoxic zones grow from diffuse and small O_2-starved zones in the coastal regions. Excess N has created vast O_2-depleted coastal areas. Crops absorb as little as 20% of the N applied, and a large fraction by runoff (Smil, 1999). The areal extent of water affected by hypoxia varies tremendously at local, regional, and continental levels in accord with area under intensive crop production. Anthropogenic activities have altered the world's N cycle and C cycles. While N supplies are extremely limited in some regions of the world, they are high in other regions and cause pollution of freshwater. For example, in Sub-Saharan Africa, about 80% of the cropland soils are nutrient-depleted with negative nutrient budget, severely degraded, and producing low yields. The annual negative nutrient balance includes about 22 kg of N and 2.5 kg of P per hectare of cropland across the African continent (Sanchez, 2002).

One of the important ecological zones in the USA that is severely affected by the hypoxia is the Gulf of Mexico which includes coastal waters of Louisiana and Texas (Fig. 14.8) (NOAA, 2003). This region is widely recognized as a "dead zone" because of the O_2-starved waters. This hypoxic zone has expanded from 20,000 km^2 in 1999 to 22,000 km^2 in 2002 and its intensity has also increased (Kaiser, 2005). In summer 1999, billions of aquatic organisms were suffocated because of the lack of O_2 in the Gulf of Mexico, which was the worst hypoxic disaster in the region (Ferber, 2001). The excessive use of N and P fertilizers throughout the Mississippi River Basin comprising the Midwestern region is the principal factor responsible for the development of hypoxic zones.

The excessive delivery of N- and P-rich waters by the Mississippi River is the main cause of the development of this hypoxic zone. The Mississippi River transports more than 1.6 million metric tons of N washed off from agricultural fields from the Midwestern states every spring (Ferber, 2001). The high N and P input favors a rapid explosion in growth of algae, zooplankton and other aquatic organisms during summer, and later the death of these organisms consumes all the O_2, dropping the O_2 levels to <low $2\ mg\ l^{-1}$ along the coast. The O_2 depletion is the highest in summer and the lowest in winter. It increases gradually from spring to summer, peaks in summer, and then decreases to a minimum in fall, mirroring the growth of algae and other aquatic plants.

Three times as much N were delivered to the Gulf in 2006 compared with that in mid 1970s (NCAT, 2006). Because fertilizers are relatively inexpensive, many farmers apply fertilizers more than (10 to 20%) what the soil needs to maintain or increase crop yields. The magnitude and rate of nutrient and sediment release from agricultural lands are, however, difficult to accurately assess because of the nonpoint source nature of pollutants. Although there is still a need for a more quantitative documentation of the amounts of N and P delivered by the Mississippi River Basin,

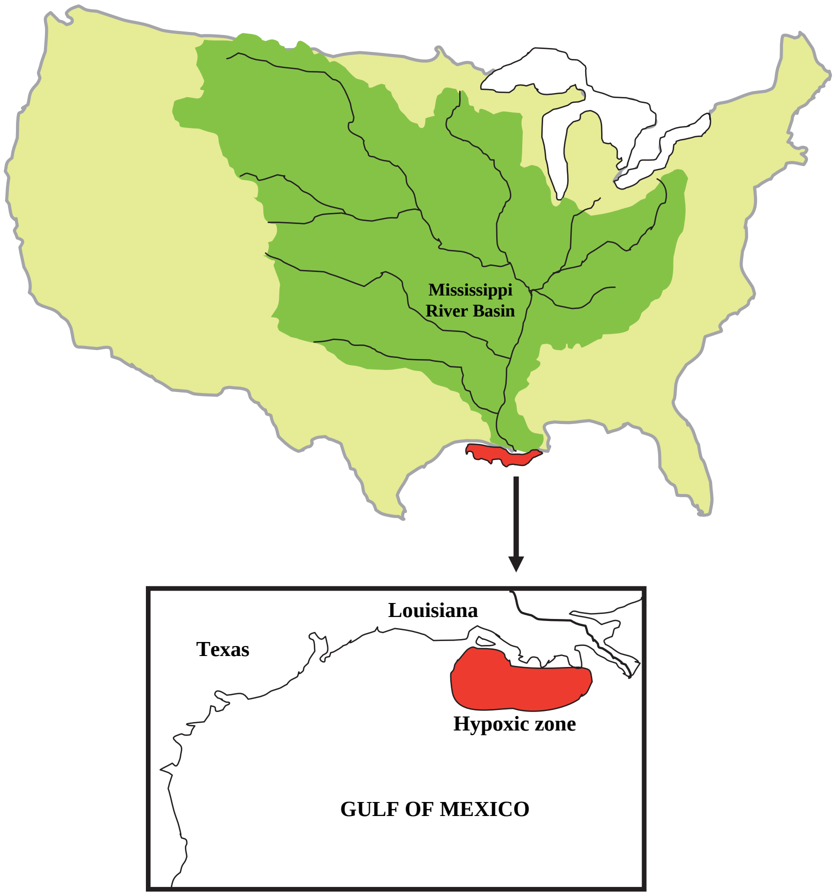


Fig. 14.8 The Mississippi River Basin and the hypoxic zone in the Gulf of Mexico (After U.S. Geological Survey, 2006; Kaiser, 2005)

there is a general consensus that heavy applications of N and P fertilizers and animal manure in interaction with intensive farming practices are the main causes of the development of hypoxia in the Gulf of Mexico. Other pollutants (e.g., municipal and industrial wastes, airborne chemicals) also contribute to the problem.

14.9 Wetlands and Pollution

A wetland is a land area that remains saturated with water most of the time and is neither completely terrestrial nor completely aquatic. The interaction of ponded ecosystem with plant and animal communities living in the saturated soil-water environment determines the type and nature of wetland development. Because wetlands

Fig. 14.9 Well-maintained wetlands trap and filter runoff sediment and improve water quality (Courtesy USDA-NRCS)

are transitional ecosystems between upland terrestrial and aquatic systems, they provide ecosystem services vital to minimizing transport of pollutants while improving water quality. Wetlands are synonymous with lungs of ecosystems, and are ecological, biological, and natural method of treating non-point source pollution or nutrient overloading (Fig. 14.9). They intercept surface runoff and subsurface lateral flow and filter pollutants, constituting a natural sink of runoff sediment and chemicals from agricultural lands and urban ecosystems which would otherwise be delivered directly and rapidly to downstream water bodies including streams, rivers, lakes, and coastal waters. The retention of runoff triggers the various physical, chemical, and biological processes of pollutant (e.g., N, P) removal, transformation, and degradation including nutrient uptake, denitrification, mineralization, immobilization, and filtration.

14.9.1 Degradation of Wetlands

Wetlands are dynamic hydrological systems, and are prone to degradation, through a range of external and internal factors which exacerbate the problem. External factors include sediment and nutrient loads in runoff mainly from agricultural lands and flooding. In comparison, internal factors include drainage, eutrophication, salinization, decomposition of organic substrates, displacement of native plant species by competitive invaders, and pest invasion. Because wetlands are located toward downstream portions of the landscape, they receive a wide range of pollutants. Flooding of wetlands and dredging and construction of levees along main water courses lead to permanent alteration of natural wetlands. The elimination of vegetation from neighboring ecosystems impairs the buffering capacity of wetlands to function. About 50% of the wetland areas in the world have been lost by conversion to agricultural and other land uses. The land area under wetlands worldwide ranges between 5.3×10^6 and 12.8×10^6 km^2, but most of the wetlands are affected by degradation (Zedler and

Kercher, 2005). This shows that less than 9% of total land is under wetlands. Although they occupy only a small portion of the landscape, wetlands are essential to reducing of non-point source pollution, increasing biodiversity recovery, controlling floods, improving water and air quality improvement, and sequestering C. Poorly managed wetlands can be a source of water and air pollution. For example, draining wetlands can increase rates of mineralization and release of greenhouse gases (e.g., CO_2, CH_4, N_2O) because of the decomposition of organic matter. Wetlands are rich in organic matter and have high redox potential.

In the USA, about 53% of wetlands were drained between 1780 and 1980, with major changes in area in Florida, Texas, Ohio, Indiana, Illinois, and Minnesota (Dahl, 1990). Wetlands occupy about 11.9% of the area of the 50 states of which Alaska represents 6.9%. The wetland area for the 48 contiguous states accounts for only 5% of the total land area in the U.S. (Zedler and Kercher, 2005). In the midwestern states within the Mississippi River Basin, between 80 and 90% of the natural wetlands have been drained for land use conversion (Mitsch and Day, 2006). Drainage of wetlands from the midwestern U.S. states contributed to increased flooding and hypoxia in the Gulf of Mexico. Expansion of agriculture, urbanization, and logging operations are among the principal factors responsible for the decrease in wetland areas in the U.S. Farmers often cultivate until the edge of farms or water bodies, reducing protective land (riparian zone) between croplands and streams that could be used for establishment of buffers or other conservation measures.

14.9.2 Restoration of Wetland

Maintenance, creation and restoration of wetlands are critical to improving water quality, promoting wildlife habitat, and protecting lands from flooding and inundation. Restoration of wetland ecosystems with native species is a viable strategy for improving the environment. In the USA, landowners often create small ponds to improve the scenery and wildlife habitat, but conversion of natural wetlands to agricultural lands is far greater than the area under created wetlands or ponds. About 2.2 Mha of wetlands in the midwest USA need to be created or restored to remove 40% of N discharge into the Gulf of Mexico (Mitsch et al., 2005). Creation of river diversion wetlands is also a potential measure to reduce transport of pollutants. A river diversion or riparian wetland is a wetland adjacent to the main channel of a river. It is created by pumping sediment- and nutrient- water or flooding from the main channel. Runoff wetlands and conservation buffers established between agricultural lands and aquatic systems are useful strategies for significant reductions in N and P transport.

14.10 Mitigating Non-point Source Pollution and Hypoxia

The way forward to addressing the problem of water pollution is by: (1) maximizing the use efficiency of N and P fertilizers, which would directly reduce excessive input of inorganic fertilizers and animal manure, and (2) adopting recommended agricultural systems that trap sediments and remove nutrients from runoff prior to reaching the water courses. Thus, avoiding excessive use of fertilizers and pesticides and implementing the best management practices are two principal options of reducing the non-point source pollution.

14.10.1 Management of Chemical Inputs

Methods, rates, timing of application of fertilizers, manure, and pesticides are important considerations to reducing adverse environmental impacts. For example, time between the application of fertilizer or manure and occurrence of the first rainfall event determines the severity of nutrient pollutant transport in runoff. Threshold levels of application of fertilizers and manure must be determined for every soil to reduce over-enrichment in nutrients. Rate of application of fertilizers and manures must be based on crop requirements. Recommended technologies (e.g., precision agriculture) offer an opportunity to apply the exact amounts of N needed by the crops.

Nutrients supplied by manuring must be accounted for in the calculation of the rate of fertilizer application. Soil tests used to develop fertilizer recommendations must be based on different mechanisms of nutrient dynamics and cycles that occur in the soil following application including volatilization, release and availability, mineralization, immobilization, transformations, solubility, sorption, desorption, precipitation (e.g., P), and dissolution. For example, soils vary in their P sorption and desorption capacities and formation of P precipitates, depending on clay amount and mineralogy. Split applications and placement in bands of fertilizers and herbicides are recommended measures to enhance uptake. Split applications in early spring and late fall reduce the rate of application and improve the efficiency.

A precise application of fertilizer and other agrichemicals and proper utilization of animal manure are required to significantly reduce the problem of non-point source pollution and of hypoxia in coastal ecosystems. It is estimated that 30% reduction of N discharge by 2015 would shrink the hypoxic zone in the Gulf of Mexico to about 5,000 km^2 (Ferber, 2001). Measures that reduce the off-site transport of pollutants from source areas can reduce the N and P loads in coastal areas. Adoption of best management practices can significantly reduce off-site migration of pollutants.

Controlling non-point source pollution requires the understanding of factors contributing to runoff including soil infiltration capacity, topography, climate (e.g., rainfall, wind, temperature), and land use and tillage systems. Because runoff is the main pollutant carrier, measures that control excessive runoff must be identified and applied to areas where fertilizers and manure have been applied. Runoff initiation depends on the nature of the soil surface at the time of rainfall or irrigation. Thus, practices that minimize runoff generation can reduce non-point source pollution. Understanding of processes and factors of runoff and pollutant transport from the source areas to receiving waters is crucial to control pollution.

Velocity of runoff is a principal driving force of water erosion. Thus, reducing runoff velocity is crucial to reducing transport of sediment and pollutants in water. Deposition occurs with decrease in velocity of water runoff. Sand or other large particles settle first followed by silt and clay particles. According to the Stoke's law, time to settle 0.5 m is 40 s for sand particles, 4 h for silt, and 24 h for clay.

14.10.2 Conservation Practices

Use of conservation tillage with innovative practices including conservation buffers, crop residue management, nutrient management, integrated pest management, winter cover crops, and other improved management technologies are effective strategies for controlling non-point source pollution. Combined use of these practices may be more effective than a single practice.

Some of the important practices for reducing non-point source pollution are briefly discussed below:

Vegetative cover. Leguminous covers crops not only reduce soil erosion but also increase soil organic matter, fix N, increase N availability to succeeding crops, and reduce the use of inorganic fertilizers, thereby reducing risks of non-point source pollution. Runoff and herbicide losses from no-till fields with hairy vetch cover crops were higher than those from fields without cover crops (Sadeghi and Isensee, 2001). A no-till system combined with proper rotations and cover covers is a valuable practice to reduce use of fertilizers and herbicides. Tillage practices and cropping systems that improve formation of soil macroaggregates promote sorption and reduce the risks of nutrient transport in runoff. In N-depleted soils, cover crops and N-fixing species (e.g., *Sesbania, Tephrosia, Crotalaria, Glyricidia, Cajanus*) can provide 100 to 200 kg N ha^{-1} (Sanchez, 2002).

Crop rotations. Crop rotations enhance organic matter pool and reduce the use of fertilizers, thereby minimizing non-point source pollution. An effective nutrient and pest management plan must include crop rotations as part of the integrated approach of reducing non-point source pollution. Effective rotations include grain crops and forages. Use of complex rotation is important because it breaks cycles of weeds while reducing use of fertilizer and protecting water quality.

Conservation tillage. Losses of nutrients in runoff are a function of tillage and cropping systems. A tillage system that leaves large amounts of residue on the soil surface generally reduces runoff and soil erosion. No-till management is particularly effective in reducing pollutant wash-off in runoff because it improves soil structure and water infiltration rate. Excessive tillage alters soil surface conditions and thus disrupts water balance (e.g. precipitation, evaporation, infiltration), increasing surface runoff. Conservation tillage systems which rely more on biological techniques than on chemicals (e.g., herbicides) for weed control can minimize concerns over non-point source pollution. Soil erosion is a selective process and preferentially carries microaggregates. Because concentrations of C, N, and P tend to be higher in macroaggregates ($>$0.25 mm) than in microaggregates ($<$0.25 mm) in no-till (Green et al., 2005), plowed soils are more prone to nutrient erosion in microaggregates than no-till soils. The length of time that a soil remains bare before and after spring tillage must be minimized, because runoff and soil erosion are the highest in spring when soil is bare and rainstorms are intense. Contour cultivation such as contour strip-cropping in combination with no-till management is another technology for reducing runoff and soil transport. It increases water infiltration because of different root patterns and canopy cover of the crops.

Crop residue. Residue mulch reduces soil detachment and sediment transport particularly during the critical times. Crop residues intercept raindrops, stabilize soil aggregates, and increase biological activity to reduce excess soluble nutrients. There exists an inverse relationship between nutrient load in runoff and percent residue cover. Losses of pollutants in runoff decrease exponentially and negatively with increase in residue cover. Grande et al. (2005) observed the following significant relationships between total P and percent residue (PR) cover in no-till corn:

$$\text{Fall manure:} \qquad \text{Total P} = 125\,\exp(-0.044^{*}\text{PR}) \tag{14.1}$$

$$\text{Spring manure:} \qquad \text{Total P} = 85\,\exp(-0.056^{*}\text{PR}) \tag{14.2}$$

$$\text{Fall manure:} \qquad \text{Dissolved P} = 0.59\,\exp(-0.01^{*}\text{PR}) \tag{14.3}$$

$$\text{Spring manure:} \qquad \text{Dissolved P} = 0.74\,\exp(-0.02^{*}\text{PR}) \tag{14.4}$$

Residue mulch has a high ability to absorb and retain chemicals, and allows the reaction of chemicals with soil and transported with sediment. As much as 50% of herbicide applied to the soil surface can be trapped by residue mulch (Selim et al., 2003). The mechanisms of herbicide adsorption by residues differ from those of soil. Although herbicides intercepted by residues can be washed off rapidly under intense rains, in some cases, their release is slow, increasing the use efficiency of herbicide.

Conservation buffers. Buffers can effectively remove pollutants from runoff when strategically installed within and at the edge of the croplands (USDA-NRCS, 2006). For example, grass barriers planted on the contour are effective means for reducing transport of pollutants from agricultural fields because they intercept and delay runoff flow and promote sediment deposition and infiltration above and within them. Vegetative filter strips also remove sediment, organic matter, and other pollutants from runoff. Any vegetation established along field borders or at the field edge reduces incoming runoff and traps sediment and nutrients. The effectiveness of buffers for retaining pollutants depends on the buffer width, source/buffer area ratio, buffer species, buffer management, soil properties, pollutant concentration and properties, and management of the source area (Krutz et al., 2005). Benefits of buffers are further discussed in Chapters 9 and 10.

Mechanical structures. Engineering or mechanical structures include terraces, containment systems, sedimentation basins, or ponds to retain and collect runoff. Well-managed structures although expensive reduce runoff and soil erosion. Their effectiveness can be enhanced when combined with improved upland practices such as contour cropping, strip cropping, conservation tillage, and buffer strips.

14.11 Models of Non-Point Source Pollution

Several empirical and process-based models have been developed since 1970s to predict the movement of chemicals in runoff. The non-point source pollution models simulate the transport and fate of agricultural chemicals and integrate losses of runoff in surface runoff as well as lateral and vertical flow components through saturated and unsaturated soil media. Examples of commonly used models are the USLE, MUSLE, RUSLE, EPIC, SWAT, WEPP, ANSWERS-2000, Annualized Agricultural Non-Point Source Pollution Model (AnnAGNPS), Chemicals, Runoff, and Erosion from Agricultural Management Systems (CREAMS), Pesticide Root Zone Model (PRZM), and GLEAMS (Parsons et al., 2004). The USLE, RUSLE/MUSLE, EPIC, and WEPP are mainly used for sediment transport predictions. Application, time scale, and spatial scale of these models vary. The models are user-defined and can accommodate plot, field, watershed, and basin scale studies. The above models can now be combined with GIS to model sediment and pollutant transport in response to varying rainfall events and contrasting management scenarios of soil and agrichemicals (Refer to Chapter 4).

Summary

The quality of surface and subsurface water resources is declining due to the high anthropogenic activities. Inappropriate agricultural systems, high concentration of livestock in small areas, and increased use of fertilizers and pesticides are some of the leading causes for pollution of water bodies. Water pollution and degradation of wetlands with runoff sediment are major concerns. Intensive soil disturbance in agricultural and non-agricultural lands has altered the soil surface conditions and increased rates of runoff and sediment losses. The two main sources of pollution are point source and non-point sources. The former refers to the water pollution that originates from known sources such as discharge pipes of wastewater from urban facilities whereas the non-point source pollution does not have a specific identifiable source. The later is the main cause for pollution of water because it can not be controlled easily. Characteristics of the pollutant source area, soil and water management, climate, and types of pollutants influence the magnitude of water pollution. Agriculture, urbanization, acid precipitation, mining, deforestation, road constructions, and other anthropogenic-induced activities are the main source of pollutants. Sediment, N, P, animal manure, and pesticides are the common pollutants, which are transported through surface runoff, subsurface lateral flow, leaching, and volatilization.

Hypoxia in coastal waters is one of the principal problems due to the excess input of N. Hypoxia is the depletion of O_2 in water, which causes eutrophication. It normally occurs when the O_2 levels drop below $2\,\mathrm{mg\,l^{-1}}$. About 150 hypoxic zones exist around the world and cover more than $70{,}000\,\mathrm{km}^2$ of surface area including the Gulf of Mexico, Baltic Sea, northern Adriatic Sea, Gulf of Thailand, Yellow Sea, and Chesapeake Bay. The Gulf of Mexico is increasingly being recognized as a “dead zone” due to the high intensity of eutrophication and the excessive use of N and P fertilizers in the upstream Midwestern region. Strategies to mitigate water pollution and development of hypoxic zones include efficient use of N and P fertilizers, animal manure, and establishment of improved soil and water conservation practices (e.g., conservation tillage, residue management, conservation buffers, contour farming, engineering structures). Now, empirical and process-based models are used to predict the movement of fate and pollutants. Models allow the simulation of the magnitude of water pollution across different time and spatial scales. The models can be also combined with geographic information systems to assess pollution sources and establish remediation strategies across regional scales.

Study Questions

1. Compare and contrast hypoxia, eutrophication, allelopathy, and non-point source pollution.
2. Discuss the major hypoxic zones in the world. Causes.
3. What are the indicators or parameters used to indicate the quality of water for irrigation.?
4. How do models contribute to water quality management.?
5. What are the mechanisms by which cover crops and crop residue mulching reduce runoff.?
6. Does no-till farming reduce chemical and water runoff in all soils? If not, why.?
7. Discuss all the pathways by which nutrients and pesticides are lost from the soil.
8. What are the main sources of pollutants.?
9. Discuss the management practices for restoring polluted ecosystems.
10. Describe pollution in wetlands.

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Chapter 15
Restoration of Eroded and Degraded Soils

Restoration refers to the process of repairing and returning damaged or degraded soils (e.g., eroded agricultural soils, mined soils) to a condition similar to the pre-degradation level of capability for supporting plant growth and maintaining environmental quality. Degradation, which broadly refers to decline in soil's capacity for a specific use (e.g., productivity, pollutant filtration, C sequestration) is somewhat a qualitative term. Thus, its quantitative parameterization is a high priority. In this textbook, the term *restoration* is used instead of *reclamation*, which sometimes refers to the processes of creating new lands.

Restoring degraded soils is an absolute necessity for:

1. meeting the increasing demands for food,
2. reducing the increasing adverse pressure on prime agricultural soils,
3. recovering the value of degraded soils and enhance wildlife habitats,
4. reducing pollution of surface and ground water resources,
5. enhancing economic development and agricultural sustainability, and
6. avoiding risks of irreversible soil degradation.

Land availability for expanding the cultivated area is finite and diminishing. Thus, the only two viable alternatives for meeting the demands for food and fiber are by: (1) increasing the crop yield per unit of area, and (2) restoring degraded/marginal or hitherto to unproductive soils. Evaluation of the extent of soil degradation, identification of degradative processes, and implementation of appropriate mitigation technologies are strategies for restoring degraded soils. A reversal may not be possible in strongly degraded soils. Severely eroded, nutrient-depleted, poorly drained, saline and sodic, compacted, and acid soils are examples of systems that require restoration. Restoration of degraded soils must adhere to the following principles: (1) increase in soil organic matter content, (2) improvement in soil structural properties, (3) management of runoff, and (4) protection of exposed soil surfaces.

The length of time required to restore degraded soils varies with ecosystem conditions, severity of degradation, restorative strategies, and climate. A degraded soil that has not lost its topsoil can recover within 3–5 yr under proper management (Daily, 1995). Severely but not irreversibly degraded soils can require as long as 100 yr to recover to a self-sustaining state suited for agriculture.

H. Blanco, R. Lal, *Principles of Soil Conservation and Management*,

15.1 Methods of Restoration of Agriculturally Marginal Soils

An agriculturally marginal soil is prone to accelerated erosion and is thus characterized by adverse properties, and low fertility and productivity. Adopting practices such as reforestation, conservation tillage (e.g., no-till, reduced tillage), crop rotations, manuring, and application of organic amendments is an alternative to restore marginal soils (Table 15.1). Because soil degradative processes (e.g., water and wind erosion, salinization) are most severe in places where there is no vegetation, introduction of trees and forage crops can protect soil surface and improve the subsoil. Soil erosion from degraded and agriculturally marginal soils (162 $Mg\,ha^{-1}$) can be as much as 50 times higher than from soils managed by under conservation effective practices (3.3 $Mg\,ha^{-1}$) (De Santisteban et al., 2006). The above biological technologies can restore partially degraded soils, but severely degraded soils require costly restoration practices.

A brief explanation of each restorative practice is presented below:

1. ***Vegetative cover***. One of the first strategies to restore agriculturally marginal soils is the establishment of vegetative cover, which includes trees (e.g., forest plantations), intercropping trees with legumes, perennial native grass species, etc. Nitrogen-fixing trees such as *Leucaena* spp. for nutrient depleted soils, and fast growing trees such as *Eucalyptus* spp. for improved drainage in waterlogged and saline soils are examples of tree management. Trees produce abundant biomass and floor litter, which increases soil organic matter and nutrient contents. Likewise, perennial grass species stabilize the soil, reduce soil erosion, and improve soil physical properties. Introducing agroforestry systems is also an effective conservation technology. Combining trees with field crops restores soil productivity because trees improve water infiltration and provide economic benefits to farmers. Multipurpose tree species, for example, provide food, wood, and fiber along traditional food crops. Establishment of conservation buffers (e.g., filter strips, riparian buffers, grass hedges, grass waterways) is part of best management practices to stabilize marginal soils and reduce off-site transport of sediment and nutrients. An appropriate use of marginal soils could generate income through restoration and enrollment in conservation programs. In the USA, programs such as the CRP and the Conservation Reserve Enhancement Program (CREP) offer an option to convert highly erodible soils (FSA, 2006). In degraded pasturelands, livestock rotation is critical for maintaining soil resilience because it allows the system to rejuvenate and recover from grazing.

Table 15.1 Some strategies to restore agriculturally marginal soils

Vegetative cover	Cropping systems	Tillage systems	Amendments
• Agroforestry	• Crop rotations	• No-till	• Crop residue mulch
• Fast growing trees	• Intercropping	• Reduced tillage	• Animal manure
• Trees and grass species	• Cover cropping	• Subsoiling	• Green manure
	• Contour farming	• Mulch tillage	• Lime
• Wetlands and recreation areas	• Organic farming	• Strip/ridge tillage	• Compost

2. ***Tillage and cropping system***. Intensive cultivation is one of the main factors responsible for soil degradation because it accelerates decomposition of organic matter and modifies soil properties. No-till system is thus a recommended practice because it reduces the rate of residue decomposition, protects the soil surface, and promotes soil aggregation and microbial activity. Introduction of cover crops, incorporation of legumes in crop rotations, intercropping, and mixed cropping are recommended strategies to increase soil organic matter content and improve soil fertility. Legume crops fix their own N from the atmosphere and reduce the use of inorganic fertilizers in nutrient-depleted soils. Both grain and forage legumes are viable alternatives for crop rotations with N-demanding crops. For example, planting soybean after corn or fallow regenerates soil fertility by enhancing proliferation of N-fixing organisms and thus increasing N levels. The improvement in N levels results in higher yields of the following crops. Cover crops improve soil resilience but are best suited to temperate and warm regions with abundant precipitation. In arid and semiarid regions, cover crops may compete or reduce the available water for subsequent crops. Diversified crop rotations with perennials forages provide a permanent cover and protect soils against water and wind erosion. Conversion of agriculturally marginal soils to permanent crops also improves wildlife habitat and enhances biodiversity. Most importantly, it can promote C sequestration and reduce emissions of greenhouse gases.
3. ***Use of crop residues and amendments***. Because degraded soils are often characterized by acidic or alkaline pH, low organic matter and nutrient contents, compacted horizons, and low water retention capacity, application of organic amendments is a desirable strategy. Degraded soils respond rapidly to use of organic and inorganic amendments. Animal manure, green manure, compost, and other nutrient-rich materials react with soil and improve fertility and biological activity. Crop residues left on the surface not only protect soil but also provide organic matter. Compost is a rich source of active soil organisms, organic matter, and nutrients that improve soil resilience (Cox et al., 2001). Applying gypsum is another effective measure to improve soil resilience. In arid and semi-arid regions, runoff and soil loss from soils treated with gypsum can be about 45% of those from untreated soils (Agassi et al., 1989). Composting or recycling yard and food wastes is also an important strategy to increase plant growth and yield while easing pressure on landfills.
4. ***Natural fallows***. Natural fallow management is a common practice to restore slightly and moderately degraded soils. This practice consists of retiring a cultivated soil for one or more growing seasons. Natural fallows rely on the intrinsic and natural ability of the soil to regain its potential under naturally grown vegetation, mostly grasses and shrubs. Restoration of soil physical properties and fertility often requires that soils be left in natural fallow for extended period of time (15 to 20 yr). Agricultural pressure has, however, necessitated the reduction of the duration of fallow to merely 2 or 3 yr in many ecosystems, jeopardizing the full regeneration of soil and vegetation. Soil recovery to an equilibrium level increases in direct proportion to the length of natural fallow. The length of natural fallow has unfortunately been based on the needs of landowners and policymakers rather than on the needs of the system to fully recover its potential.

Differences in resilience among soils determine the optimum duration of natural fallow periods. Long-term and site-specific studies are critical to determining the optimum time required for the soil to recover. Severely degraded ecosystems may require long (>20 yr) fallow periods. In mountainous tropical ecosystems, the use of natural fallows is linked to shifting cultivation where cultivated soils (often degraded) are left in fallow to recover their natural fertility while neighboring soils are brought under cultivation by slash and burn system. Introduction of fast growing trees, shrubs, and grasses accelerate the soil recovery process.

5. ***Nutrient and pesticide management***. One of the key roles of restoring degraded soils is to reduce the environmental pollution by filtering, detoxifying, and degrading excess chemicals. Because soils can become polluted, management of pesticides and fertilizers is crucial. While pesticides and fertilizers are important to agricultural production, their excessive use pollutes soil, water, and air. Applying only the necessary amounts of chemicals and monitoring the soil following application are important measures. Use of non-chemically based practices (e.g., organic farming, crop rotations, cover crops, manure management) is desirable to reduce non-point source pollution. Proper management of animal manures and bio-solids is also important to protecting the water quality. Excessive application of animal manure pollutes surface water with pathogens, ammonia, soluble N and P, and other organic materials.

15.2 Compacted Soils

Compaction refers to the process of densification caused by close packing of soil particles due to external compressive forces that create a dense soil material with reduced total and macroporosity. Soil compaction is probably one of the most severe degradative processes in mechanized agriculture. The advent of heavy tillage machinery under highly mechanized agriculture and animal traffic (e.g., overgrazing) from concentrated livestock farms are the causes of soil compaction. Repeated wheel traffic during tillage, planting, harvesting, manuring, and weed and pest control degrades the soil structure and causes surface and subsurface compaction in cultivated soils. Subsurface compaction occurs when a compacted subsurface horizon of higher bulk density and lower total porosity than the topsoil, known as plowpan, is formed (SSSA, 2006). Extent of compaction depends on axle load, tillage methods, and site-specific conditions (i.e. texture, drainage). Soil compaction is particularly a major challenge in poorly drained clayey soils with high shrink-swell potential (Flowers and Lal, 1999).

Compacted soils are characterized by high bulk density, penetration resistance, and shear strength. These adverse soil physical properties restrict root growth, limit plant emergence, and reduce crop yields. High levels of soil compaction also reduce water infiltration rate and increases risks for water pollution. On a claypan silt loam, wheel traffic reduced saturated hydraulic conductivity and increased runoff in continuous cultivated fallow plots (Blanco-Canqui et al., 2004). Wheel trafficked zones in agricultural fields normally have higher runoff and soil erosion rates than non-wheel trafficked zones. Wheel tracks often generate risks of concentrated flow. Strategies for managing compacted soils include:

1. ***Subsoiling***. Subsoiling, although expensive, is a common practice to loosen severely compacted soils. It shatters the plowpan and improves soil internal drainage. The benefits of subsoiling can be transient due to risks of recompaction. Thus, subsoiled fields must be maintained under controlled traffic.
2. ***Mulching.*** Residue mulching and application of green or animal manure are also effective measures to reduce soil compaction. Coarse and dense mulch cover buffers compactive forces or stresses. Organic mulches also promote proliferation of beneficial earthworms and microorganisms which improve soil structure and water infiltration rate of compacted soils.
3. ***Cropping systems***. Crops that leave all residues after harvest minimize the effects of compaction forces. Rotations that include deep-rooted and N-fixing crops such as alfalfa, clover, and sunflowers are viable alternatives to manage soils compacted by monocropping with shallow rooted crops.
4. ***Controlled traffic***. Because compacted soils can be difficult to recover, preventing compaction is crucial. Traffic must be minimized and confined to permanent lanes. Timing of vehicular traffic determines compaction the severity of soil compaction.
5. ***Soil wetness***. Wet and bare soils become more easily compacted than dry soils under the same amount of force. Even a single wheel traffic pass can cause dramatic compaction if the soil is wet. Characterizing plastic and liquid limit attributes of a soil is essential to estimating the ease with which a soil becomes compacted. Traffic must be performed whenever the soil is at or drier than the plastic limit. Soils with good tilth can allow traffic at higher water contents than those with poor tilth.
6. ***Axle loads and tire pressures***. Reduction of axle loads is the key to decreasing excessive soil compaction. High tire pressure compacts surface soil layers more than low tire pressure because the latter spreads out the compaction forces on the soil (Petersen et al., 2006).
7. ***Reduced tillage***. Plowing destroys the natural soil aggregates, eliminates biopores, and reduces pore size. It thus increases the soils' susceptibility to compaction. Reduction of primary tillage and elimination of secondary tillage operations decrease risks of soil compaction. A permanent ridge-till system is a desirable practice to confine traffic to fixed paths.

15.3 Acid Soils

Acid soils occupy between 30 and 40% (1.5 billion ha) of the total arable land area in the world (Herrera-Estrella, 2003). These soils mostly occur in highly weathered and leached tropical and temperate environments. Ultisols are most severely affected by acidity (Fig. 15.1), and are predominant in tropical regions

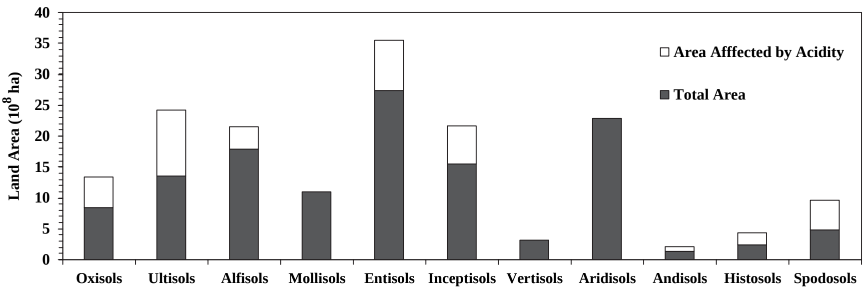

Fig. 15.1 Estimated land area affected by acidity within each soil order (After Sumner, 1998)

of southeast Asia, Africa, and South America (Haynes and Mokolobate, 2001). Excessive use of N fertilizers, high rates of N leaching, and acid rain are the main causes of soil acidification. Ammoniacal fertilizers through nitrification release NO_3^-, which in reaction with H^+ forms strong acids. Formation of acids depends on the rates of N uptake by plants, volatilization, nitrification, and soil and climatic conditions.

Strongly acid soils (pH<4.5) often have degraded structure and poor nutrient uptake and crop yields. Acid soils are characterized by a high concentration of H^+ resulting from the excessive leaching of basic cations (e.g., Ca^{2+}, Mg^{2+}). They also possess high levels of soluble Al and Mn. In neutral soils, Al^{+3} and Mn^{+3} commonly occur in insoluble forms (e.g., aluminosilicates), but they become soluble in strongly acid soils. Toxicity of Al and Mn inhibits nutrient uptake and growth of plant roots. Prevalence of sesquioxides (Al, Mn, and Fe oxides) strongly adsorbs P and thus reduces its availability (Haynes and Mokolobate, 2001). Soil salinization is common to arid and semi-arid regions, whereas soil acidification is common to areas with high rainfall (>500 mm), and, thus, high rates of leaching.

15.4 Restoration of Acid Soils

Liming, P fertilization, addition of organic amendments, and introduction of acid-tolerant crops or plant species are some of the viable practices for restoring acid soils. Liming is the most commonly used method of correcting acidity. The pH levels, Al^{3+} and Mn^{2+} toxicity levels, clay content, clay mineralogy, and organic matter content determine the rate of lime application. Soil specific lime requirement curves must be constructed based on the soil pH response to liming. Lime chemically reacts with the soil water and produces Ca^{2+} and OH^-. The Ca^{2+} replaces the Al^{3+} and H^+ in the exchange complex sites, and the released hydroxyl ions (OH^-) react with Al^{3+} and H^+ and form H_2O and insoluble $Al(OH)_3$ as follows:

$$CaCO_3 + H_2O\ (\text{soil water}) \Rightarrow Ca^{2+} + 2OH^- + CO_2\uparrow$$

$$2Ca^{2+} + \text{Soil}\ (Al^{3+} + H^+) \Rightarrow \text{Soil}\ (2Ca^{2+}) + Al^{3+} + H^+$$

$$Al^{3+} + 3OH^- \Rightarrow Al(OH)_3 \text{ (solid)}$$
$$H^+ + OH^- \Rightarrow H_2O$$

Example 1. How much calcitic limestone is needed to restore an acid soil that has a pH of 4.8, base saturation of 45%, and a CEC of 12 $\text{cmol}_c\ \text{kg}^{-1}$. The desired pH is 7.0 with a base saturation of 95%. The calcitic limestone has 15% of impurities.

Solution:

1) Increase in pH from 4.8 to 7.0 signifies a corresponding increase in base saturation from 45 to 95%. Thus, a 50% change in base saturation is required to correct the excessive acidity. The corresponding amount of Ca in $\text{cmol}_c\ \text{kg}^{-1}$ and in g per kg of soil is

$$\text{Amount of Ca in cmol}_c\ \text{kg}^{-1} = \frac{12\ \text{cmol}_c}{\text{kg of soil}} \times \frac{50}{100} = 6\ \text{cmol}_c\ \text{kg}^{-1}$$
$$\text{Amount of Ca in g per kg of soil} = \frac{6\ \text{cmol}_c}{\text{kg of soil}} \times \frac{40}{2\ \text{gCa/mol}_c} \times \frac{1\ \text{mol}_c}{100\ \text{cmol}_c}$$
$$= 1.2\ \text{g Ca kg}^{-1}$$

2) Since Ca^{2+} will be applied as $CaCO_3$, the 1.2 g Ca kg^{-1} must be converted to equivalent weight of $CaCO_3$

$$\text{Weight of CaCO}_3 = 1.2\ \text{g Ca kg}^{-1} \times \frac{100\ \text{g Ca CO}_3/\text{mol}_c}{40\ \text{g Ca/mol}_c} = \frac{3\ \text{g CaCO}_3}{\text{kg}}$$
$$= \frac{3\ \text{kg CaCO}_3}{\text{Mg}}$$

3) The total soil weight per ha is

$$\text{Weight of soil} = \text{Bulk density} \times \text{Depth} \times \text{Area}$$
$$= 1.30\ \text{Mg m}^{-3} \times 0.40\ \text{m} \times \frac{10000\ \text{m}^2}{ha} = 5200\ \text{Mg ha}^{-1}$$

4) The amount of pure $CaCO_3$ required to correct the acidity is

$$\text{Weight of CaCO}_3 = \frac{3\ \text{kg}}{1\ \text{Mg}} \times 5200\ \text{Mg ha}^{-1} = 15600\ \text{kg ha}^{-1} = 15.6\ \text{Mg ha}^{-1}$$
$$\text{Pure CaCO}_3 = 15.6\ \text{Mg ha}^{-1} \times \frac{100}{85} = 18.3\ \text{Mg ha}^{-1}$$

Crop residues, green and animal manure, and compost are economical alternatives to liming and P fertilization for restoring acid soils. Application of green and animal manure lowers the Al^{3+} and Mn^{2+} toxicity and increases the P levels. Decomposition of added organic materials releases a wide range of organic compounds (e.g., humic substances), which react with the soil solution and fix Al, reducing the toxicity of Al and Mn levels. The soil pH also increases with addition of organic amendments particularly within the first few months following application. For example, application of 20 Mg ha^{-1} of residues can increase pH by about 0.2–0.6 units while 40–50 Mg ha^{-1} increase it by about 0.8–1.5 units (Haynes and Mokolobate, 2001). The temporary increase in pH with addition of organic residues can increase crop yields and, most importantly, buy time before long-term ameliorative measures of acidity are established. Selection of acid-, Al-, and Mn-tolerant plant species through traditional techniques and genetic engineering must be pursued to manage acid soils. Transgenic plant species that produce organic acids reduce the formation of soluble Al^{3+} and Mn^{2+}. Ammonium-based fertilizers (e.g., mono-ammonium phosphate, ammonium sulfate) release H^+, and thus the reduction in their use increases pH.

15.5 Saline and Sodic Soils

Salinization of soils in arid and semi-arid regions with precipitations below 500 mm yr^{-1} and high evaporation rates is another major cause of soil degradation (Fig. 15.2). Flood irrigation is a major cause of salt accumulation in croplands. Saline soils have an electrical conductivity (EC) of the saturated extract $\geq$4 dS m^{-1}, sodium adsorption ratio (SAR) $<$13, and pH $<$8.5, while saline-sodic soils have an EC $\geq$4 dS m^{-1}, SAR$>$13, and pH $>$8.5, and sodic soils have an EC $<$4 dS m^{-1} and SAR$\geq$13 (SSSA, 2006). The SAR is computed using (Brady and Weil, 2002)

Fig. 15.2 Irrigation with water of low quality can cause salinization and sodification of soils (Courtesy USDA-NRCS)

$$\mathrm{SAR} = \frac{[\mathrm{Na}^+]}{\sqrt{\frac{1}{2}\left([\mathrm{Ca}^{2+}] + [\mathrm{Mg}^{2+}]\right)}} \tag{15.1}$$

The extent of saline and sodic soils varies by region. Countries with large area under saline soil ($>5\times10^6$ ha) include Russia, Argentina, China, India, Paraguay, Indonesia, Pakistan, Ethiopia, United States, and Bolivia (Szabolcs, 1989). Globally, the largest sodium affected soils occur in Australasia and North and Central Asia (Fig. 15.3). Combined effects of limited water availability and high evaporation are responsible for the accumulation of soluble salts at or near the soil surface. Excessive evapotranspiration increases upward flow of ground water by capillarity, transporting salt ions to the soil surface. About 50% of the irrigated lands are affected by salinization, and there is an increasing trend due to the expansion in irrigated lands (Keren and Ben-Hur, 2003).

Salinization normally occurs when evaporation exceeds precipitation (Fig. 15.4). The high evaporation rates reduce the amount of water available for leaching and cause accumulation of salts. Accumulation of salt powder or crusts on the soil surface is a sign of excessive salinization. Development of well-defined compact and dark layers below the soil surface due to high concentrations of sodium carbonate indicates sodification. The sodic horizons result from illuviation of exchangeable Na^+, clay, and organic particles. High concentrations of exchangeable Na^+ react with clay particles and cause swelling and dispersion of soil, closing off macropores and reducing water and air permeability. Clay and waterlogged soils are more susceptibility to sodification. Nearly 950 Mha of cultivated soils are affected by high salt concentrations from irrigation, representing 33% of the potentially arable land area of the world (Eswaran et al., 2001). For example, in Australia, about 30% of total land area is highly sodic and about 50% exhibits some sodification (Keren and Ben-Hur, 2003).

The high concentrations of salt change the soil biophysical properties, reduce water and nutrient uptake by pants, and reduce crop productivity. Salinity levels as low as $4\,\mathrm{dS\,m^{-1}}$ can reduce growth of sensitive plants while levels $\geq 8\,\mathrm{dS\,m^{-1}}$ affect

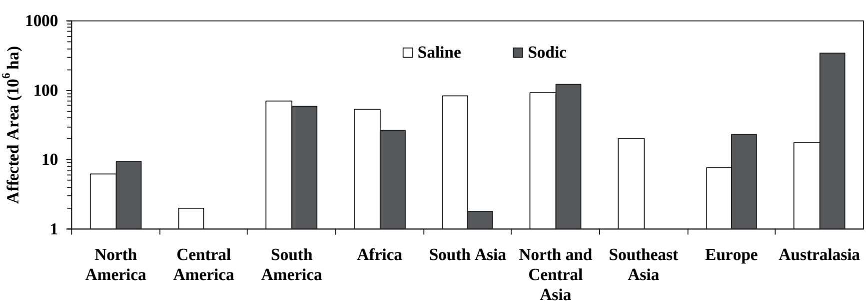

Fig. 15.3 Land area under by saline and sodic soils in the world [After Szabolcs (1989) and Rengasamy (2006)]

Fig. 15.4 Salt accumulates on the soil surface as the salt-laden irrigation water evaporation (Courtesy USDA-NRCS)

tolerant plants. High levels of salt concentrations cause partial or complete loss of crop yields. The thresholds levels of salinity are a function of cropping system, soil type, topography, and climate. Saline soils contain high concentrations of exchangeable ions of Ca^{2+}, Mg^{2+}, and K^{+}, whereas sodic soils have high concentrations of exchangeable Na^{+}. Common anions in saline and sodic soils are Cl^{-}, SO_4^{2-}, HCO_3^{-}, CO_3^{2-}, and NO_3^{-}.

15.5.1 Causes of Salinization and Sodification

Soil salinization is caused by natural and anthropogenic factors (Rengasamy, 2006). Natural factors include:

- low precipitation,
- high evaporation rates,
- deposition of wind-transported materials,
- poor drainage,
- low soil permeability,
- salt migration by capillarity from shallow water tables,
- weathering of parent material,
- gradual accumulation of salt ions from rainwater, and
- presence of lowland basins with parent materials rich in salt content and clay fractions

Depressed landscape areas collecting water and soluble compounds from surrounding upland positions are more prone to salinization. Anthropogenic factors include deforestation, overgrazing, intensive agriculture, excessive irrigation without proper drainage, irrigation with water of poor quality, application of fertilizers and organic amendments (e.g., animal manure) with high salt concentration.

15.5.2 Salinization and Soil Properties

Increase in salt concentration changes soil properties, particularly soil hydraulic properties (Levy et al., 2005). For instance, saturated hydraulic conductivity decreases rapidly with increase in the percentage of exchangeable Na^+. High concentrations ($\geq 10\,\text{mmol}\,L^{-1}$) of Na^+ cause dispersion of soil aggregates and swelling of clays, resulting in the closure of water conducting meso- and macro-pores. Soils high in montmorillonitic-clays are more susceptible to defloculation and swelling under high electrolyte concentration (Keren and Ben-Hur, 2003). The decrease in hydraulic conductivity increases surface runoff and reduces drainage. Increase in sodicity levels not only decreases hydraulic conductivity, water infiltration, and soil water retention capacity but also reduces soil structural stability, microbial biomass, and enzymatic activities. The magnitude of the negative impacts of high Na^+ concentrations depends on site specific (e.g., soil, management) conditions (Wu et al., 2003).

15.6 Restoration of Saline and Sodic Soils

Some of the common practices for restoring or managing salt- or sodium-affected soils include:

- establishing adequate drainage systems,
- leaching salts from the root zone,
- increasing high soil water content,
- subsoiling,
- applying amendments, and
- growing salt-tolerant crop species.

The choice of each practice depends on the severity of the problem, specific soil characteristics, climate, and water availability. Restoration of saline and sodic soils requires the understanding of outputs and inputs in relation to root zone salt balance (RZSB) as discussed by Qadir et al. (2000).

Change in salt storage = Inputs – Outputs = Gains - Losses

$$\Delta \text{RZSB} = (I + G + R + W + A + E) - (L + P + H) \qquad (15.2)$$

where I is salt concentration in irrigation water, G is salt concentration in groundwater moving upward by capillarity, R is salt concentration in rainwater, W is salt derived from weathering of parent materials, A is input of salt through fertilizers and amendments, E is salt transported by water and wind erosion, L is amount of salt leached out of the root zone, P is amount of salt precipitates below the root zone, and H is amount of salt removed with crop harvest.

Modern mapping and geostatistical techniques are promising tools for identifying salt-affected areas and assessing salinity levels across regions. Remote sensing and GIS maps are innovation tools to identify, classify, map, and monitor salt-affected soils (Farifteh et al., 2006). Similarly, solute transport models are used to determine the profile distribution of salt movement. Integration of information from groundwater solute transport models with that from remote sensing, GIS, and soil survey maps enhance understanding of above- and below-ground salt flux dynamics at different scales. Mathematical leaching models (e.g., fuzzy modeling) are also useful to characterize spatial and temporal variability of soil-profile salt distribution (Metternicht, 2001).

15.6.1 Leaching

Leaching salts from the root zone is a proven method for controlling salinity. It consists of applying an additional amount of water during regular irrigation periods to drain salts and maintain salinity within the permissible levels (EC $<4\,\text{dS m}^{-1}$ and SAR <13). Leaching is most effective in leveled soils and when done before the start of rainy season to keep soluble salts below the root zone. The fraction of the total amount of water, known as leaching fraction (*LF*), required for leaching of dissolved salts is computed as (U.S. Salinity Laboratory Staff, 1954; James, 1993)

$$LF = \frac{D_d}{D_i} = \frac{EC_i}{EC_d} \tag{15.3}$$

where D_d is water depth that must drain from the root zone (cm), D_i is water depth penetrating the soil (cm), EC_i is water EC penetrating the soil (dS m^{-1}), and EC_d is water EC leaving the root zone (dS m^{-1}). In most soils, application of 50 cm of water removes about 80% of salts from an equivalent 50 cm depth of soil although the salt removal effectiveness is a function of site-specific characteristics, namely soil texture, permeability, water table depth, subsurface drainage, and the salinity level of the irrigation water (James, 1993). Leaching is not an effective strategy at removing soluble salts in soils with low permeability and shallow water tables.

Example 2. Estimate the total amount of irrigation water required to meet both the crop and leaching requirements. The irrigation requirement for the crop is 40 cm. The CE is $0.8\,\text{dS m}^{-1}$ for the water entering the soil and $4.5\,\text{dS m}^{-1}$ CE for water draining below the root zone.

Solution:

$$\mathrm{D_i} = \text{Crop water} + \text{Drainage water} = 40\,\text{cm} + \mathrm{D_d}$$

Compute $\mathrm{D_d}$ using Eq. (15.3)

$$
\begin{aligned}
D_d &= D_i \frac{EC_i}{EC_d} = (40 + D_d)\frac{0.8}{4.5} = 7.11 + 0.178D_d \\
{}^{-}D_d &= 7.11 + 0.178D_d \\
D_d &= 8.65\,\text{cm} \\
D_i &= 40 + D_d = 40 + 8.65 = 48.65\,\text{cm.}
\end{aligned}
$$

These results show that about 8.7 cm of extra water is needed to leach out soluble salts from the root zone, assuming proper performance of drainage systems and field borders (e.g., soil berms).

The method of water ponding during leaching can be *continuous* or *intermittent* (Qadir et al., 2000). *Continuous ponding* of water at the soil surface leaches the salts much quicker than the *intermittent ponding*, but it uses more water. In water deficient regions, the intermittent or split application is more convenient than continuous application because it saves 30 to 40% of water. An alternative to ponding methods is *sprinkling* in which water moves through an unsaturated media. The sprinkler irrigation is more expensive than ponding methods, but it reduces bypass flow and favors water movement through the soil matrix. The unsaturated flow slowly drains salt from the intra-aggregate sites that hold soluble salt.

15.6.2 Increasing Soil Water Content

Salt concentration and soil water content are negatively correlated. Concentrations of soluble salts decrease with increase in soil water content because of the dilution mechanism. High evaporation reduces the soil water content and thus increases salt concentrations. In soils where crops must use water with relatively high salt concentrations, maintaining high water content is an option. This can be achieved by increasing the frequency of irrigation. Trickle irrigation is desirable over sprinkler irrigation when salt content in water is high (James, 1993).

15.6.3 Use of Salt-Tolerant Crop Varieties

Using crop varieties or hybrids tolerant to saline and sodic conditions (halophytes) is a useful option to combat the negative effects of salinization. Although crops such as barley and rice can tolerate moderate salinity, most crops, especially legumes, are highly sensitive to salinity during the early stages of growth. Thus, developing crop genotypes tolerant to high salinity levels is a priority as an alternative to costly measures. Salt-resistant crops must be identified for each soil based on local information on evapotranspiration, radiation, temperature, and water content regimes. Natural and direct selection of tolerant crops from saline environments using gene mapping techniques or molecular markers are viable approaches for identifying salt-tolerant crops. Improved approaches include development of transgenic plants with genetic combinations that affect the phenotypic characteristics (Yamaguchi and

Blumwald, 2005). Use of mixed cropping systems with salt-tolerant crop species, perennials grasses, and trees can reduce accumulation of electrolytes. Biological control or phytoremediation is a growing area of research for managing saline soils (Qadir et al., 2001).

15.6.4 Use of Salt-Tolerant Trees and Grasses

Growing salt-tolerant trees and grass species is another biological, ecological, and economical alternative for managing saline and sodic soils. Deep-rooted trees and grass species improve the structure and hydraulic properties of salt-affected soils and have many advantages over other traditional restoration practices (e.g., excessive leaching). Dense and permanent trees and tall native vegetation absorb salts from the soil and use large volume of water, lowering the water table and reducing salt build-up within the root zone. In saline waterlogged soils in India, gray sheoak provided the highest yield at 20 $dS\,m^{-1}$ followed by swamp oak and beach sheoak, and that the salinity level decreased (from 25 to 12 $dS\,m^{-1}$) as height and diameter of the trees increased (Tomar and Gupta, 2002). In some soils, Kallar grass, Bermuda grass, and deep-rooted plant species such as alfalfa are used to restore salt-affected soils. In the Indus Plains of Pakistan where about 3×10^6 ha of saline-sodic soils exist, Kallar grass and sesbania reduced soil EC from 10 to 4.5 $dS\,m^{-1}$ within the 0- to 15-cm depth across three soils, and both species were more than or as effective as gypsum in reducing salinity (Qadir et al., 2002). A successful tree establishment requires the preparation of ridge trenches and drainage channels. Afforestation is a possible option of desalinization, but in highly saline soils, it must be accompanied by drainage and leaching practices. Use of salt-tolerant species can be as effective as chemical amendments or even better, depending on the salinity level and soil type.

15.6.5 Establishment of Drainage Systems

Poor soil drainage and shallow water table limit the effectiveness of salt removal by leaching. Artificial drainage systems are designed to leach out salts below the water table level and reduce the upward movement of soluble salts with the rise in water table after leaching. A drainage system consists of either ditches/channels or drainage tiles. Treatment and evaporation are some of the useful techniques to dispose saline waters collected at the end of drainage systems.

15.6.6 Tillage Practices: Subsoiling

Subsoiling, a practice where soil is plowed to about 50 cm depth to break compact layers, increases water infiltration rate, permeability, and drainage, and thus it allows an effective and rapid leaching of soluble salts to deeper layers. Subsoiling with inversion before leaching, although not suitable in no-till systems, is another useful practice to bring the subsurface soil layers with low salt concentrations to top by turning the soil over. Highly saline or sodic soils require intense forms of restoration (e.g., continuous water ponding, deep tillage, drainage tiles).

15.6.7 Application of Amendments

Application of organic amendments can also increase Na^+ leaching by improving soil macroporosity and water infiltration. On a saline clay loam in Spain, application of 14–20 Mg ha^{-1} of compost of cotton-gin residue and 5–10 Mg ha^{-1} of poultry manure over a 5-yr period decreased soil EC from 9.1 to 0.8–2.0 dS m^{-1} (Tejada et al., 2006). Application of crop residue mulch lowers the soil temperature during the day, reduces evaporation, and promotes soil water storage important to reduce salt concentrations. In some cases, coarse materials such as sand are added to the soil to enhance water infiltration and thus leaching.

15.6.8 Application of Gypsum

While the restoration techniques for saline soils can also apply to ameliorate sodic soils, there are additional specific restoration strategies for Na^+-affected soils. The common strategy is to apply gypsum (Fig. 15.5), which is rich in soluble Ca^{2+} (Qadir et al., 2001). The Ca^{2+} replaces the Na^+ in the cation exchange complex, and the released Na^+ is removed from the root zone by leaching through addition of

Fig. 15.5 Liming is a strategic measure to correct acid soils (Courtesy USDA-NRCS)

extra irrigation water. In the clay exchangeable sites, two ions of Na^+ are replaced by one ion of Ca^{2+} as follows

$$\boxed{\text{CLAY}}\begin{matrix}\diagup \text{Na}\\ \diagdown \text{Na}\end{matrix} \;+\; Ca^{2+} \Longleftrightarrow \boxed{\text{CLAY}}\text{-}Ca^{2+} + 2Na^{+}$$

The effective removal of Na^+ by leaching depends on the amendment-soil contact, soil permeability and the availability of drainage practices. Addition of Ca-based compounds improves near-surface soil physical quality and reduces surface sealing and crusting apart from replacing/neutralizing the Na^+. It flocculates soil particles, promotes aggregation, stabilizes aggregates, and increases both water infiltration rate and hydraulic conductivity (Fig. 15.6).

Other Ca-based amendments are calcium chloride, fertilizer by-products (e.g., phosphogypsum), lime or calcite ($CaCO_3$), and solid and liquid Ca fertilizers (e.g., Ca-nitrates, Ca-sulphates, Ca-phosphates). Calcium is the fifth most common element in the soil; however, most of it occurs in insoluble form such as in calcites. Thus, chemical amendments including hydrochloric acid (HCl), sulphuric acid (H_2SO_4), sulphur (S), iron ($FeSO_4.7H_2O$), and aluminum sulphate ($Al_2(SO_4)_3.18H_2O$) are used to solubilize and mobilize Ca in sodic soils (Qadir et al., 2001). Use of H_2SO_4 at low concentrations in irrigation water is also an effective practice to release Ca^{2+} from $CaCO_3$ through the following reaction (Qadir et al., 2001):

$$H_2SO_4 + CaCO_3 \Longleftrightarrow Ca^{2+}(\text{exchangeable}) + SO_4^{2-} + CO_2 + H_2O$$

Example 3. Estimate the amount of gypsum, calcite, and sulphur needed to replace the Na^+ from a saline-sodic soil of bulk density of $1.45\,\text{Mg m}^{-3}$ for a total depth of 30 cm. The soil contains 12 meq exchangeable $Na^+/100$ g of soil.

Solution:

(1) Compute the amount of Na^+ in g based on the equivalent weight of Na^+.

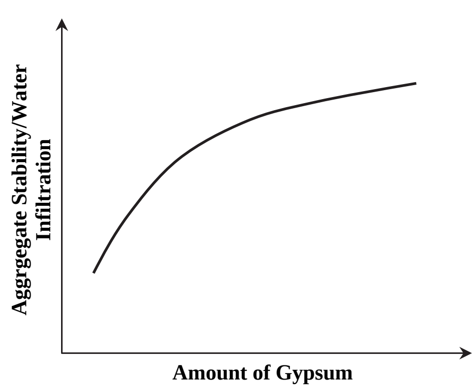

Fig. 15.6 Soil aggregate stability and rate of flow through increase with increase in the rate of gypsum application

$$\text{Amount of Na}^+ = \frac{12\text{ meq}}{100\text{ g of soil}} \times 23\,\text{mg meq}^{-1} = \frac{276\,\text{mg}}{100\,\text{g of soil}} = \frac{0.276\,\text{g Na}^+}{100\,\text{g of soil}}$$

(2) Compute the total weight of soil per ha and the total amount of Na^+ per ha.

$$\begin{aligned}\text{Total amount of soil} &= \text{bulk density} \times \text{soil depth} \times \text{Area}\\ &= 1.45\,\text{Mg m}^{-3} \times 0.30\,\text{m} \times 10000\,\text{m}^2\ \text{ha}^{-1}\\ &= 4350\text{ Mg ha}^{-1}\\ \text{Amount of Na}^+\text{ in soil} &= \frac{0.276\,\text{g Na}^+}{100\text{ g of soil}} \times 4350\text{ Mg ha}^{-1} = 12\text{ Mg ha}^{-1}\end{aligned}$$

(3) Compute the amount of gypsum, calcite, and sulphur per ha based on the equivalent weights.

$$\text{Amount of gypsum (CaSO}_4.2\text{H}_2\text{O)} = 12\text{ Mg ha}^{-1} \times \frac{172}{2 \times 23} = 44.9\text{ Mg ha}^{-1}$$

$$\text{Amount of calcite (CaCO}_3\text{)} = 12\text{ Mg ha}^{-1} \times \frac{100}{2 \times 23} = 26.1\text{ Mg ha}^{-1}$$

$$\text{Amount of S} = 12\text{ Mg ha}^{-1} \times \frac{32}{2 \times 23} = 8.3\text{ Mg ha}^{-1}$$

The estimated amounts must be corrected for the percentage of impurities of each Ca-amendment as per chemical analyses.

15.6.9 Other Techniques

Flushing and physical removal of salt-affected surface soil layers are additional techniques. Flushing the soil surface with water removes the salt accumulations from slowly permeable soils. This method is inappropriate in nearly level soils with limited gradient for water to run off. Scrapping off the salt-affected topsoils is an effective practice in small fields. It provides a temporary solution because it only eliminates surface salt accumulations leaving the salt below the soil surface unaffected. It exposes subsurface horizons, which often have lower permeability and higher clay content than the removed layers.

15.7 Mined Soils

While mining of coal and metals is essential to the global economy, it causes drastic soil and ecosystem disturbances, leading to severe soil and environmental degradation. Deforestation during mining and post-mining processes such as landslides, mudslides, water erosion, flooding, salinization, and pollution of downstream waters

Fig. 15.7 Unrestored spoil piles showing marks of gully erosion (Courtesy H. Blanco)

with heavy metals and chemical spills degrades mined soils and neighboring ecosystems (Fig. 15.7). These degradation processes, if not properly controlled, can irreversibly degrade fragile ecosystems and damage environmental quality. Expansion of the economy has intensified needs for extraction of minerals, resulting in increased soil degradation particularly in developing countries. Developing countries provide most of metals to developed countries (FAO, 2002).

The relatively unexplored but fragile mountains and highlands of several Latin American countries are among the most recently affected terrestrial systems by mining. Soil, water, and air pollution, biodiversity loss, and elimination of vegetative cover are principal concerns in mountainous mined sites. Landslides in the mountainous mined sites in China, India, and Latin America (e.g., Bolivian highlands), and pollution of rivers and streams in the Appalachian Mountains in the USA are examples of environmental degradation by mining. Wastes and excess water pumped out of mined sites contain toxic acids, heavy metals, and other pollutants. Levels of contaminants such as arsenic in mine wastewater can be about 1000 times of the standard levels (FAO, 2002). Air pollution with toxic gases, dust, and acid rain in combination with acid drainage also contributes to degradation.

Surface and underground mining are the two most common techniques. Strip mining, open-pit mining, mountaintop removal, and dredging are forms of surface mining. Surface mining often consists of removing the vegetation and excavating the earth's surface until the mineral ore is reached. Land topography, geologic material, and depth to ore determine the choice of mining method. Surface mining has a more severe impact on soil degradation than underground mining because it drastically changes the landscape geomorphology, natural vegetation, soil profile development, soil hydrology, and ecosystem function. Both surface and underground mining generate large amounts of overburden material to gain access to the desired depths. Excavated materials from underground mining deposited on the earth's surface also change the natural landscape settings and generating pollutants (e.g., sediment, toxic metals). The extent of soil degradation by mining depends on the amount of soil removed or excavated, type of minerals, topography, and climate. Coal mining is an important energy source for electric power, manufacturing, domestic heating, and transportation. Important reserves of coal are found in Australia, China, India, South Africa, and the United States, providing nearly 22%t of the world's energy and generating about 40% of global electricity (EIA, 2004). About 25% of world's coal reserves occur in the USA, and the common technique used for coal extraction is surface mining (ELC, 2002). The estimated total area permitted for coal mining in the USA is about 3.2 Mha (Office of Surface Mining, 2004).

15.8 Restoration of Mined Soils

Restoration of mined sites through adoption of proper practices is a priority to protect disturbed ecosystems. Not all abandoned mined soils around the world have been properly restored, causing concerns of environmental pollution, and ecosystem deterioration (Fig. 15.8). Restoring disturbed sites is essential to cultivation, pasture, urbanization, wildlife habitat, and recreational areas. A systematic restoration using proper management techniques permits the system to regain the soil fertility and ecosystem functionality. In the USA, the Surface Mining Control and Restoration Act passed in 1977 mandates that mining sites must be restored to their original or antecedent condition (e.g., topography, contour). While the Restoration Act has

Fig. 15.8 Expansion of mining operations in the rainforest accelerates deforestation (Courtesy Rhett A. Butler)

significantly reduced environmental pollution from the current and abandoned mining sites, pollution of water streams with heavy metals particularly from abandoned mines is still a major concern (Steen, 2006). In the USA, abandoned mined sites occupy about 560,000 ha in total (USGAO, 1996). Before 1977, the overburden material was often dumped in low lying areas where pyrite and other spoil minerals would form strong acids and create acid drainage.

15.8.1 Soil Restoration Practices

A wide range of practices is available within the portfolio of options for restoring mined soils (Table 15.2). Some of these practices are similar to those used for restoring agriculturally marginal soils (Table 15.1). Backfilling and closing mine openings from underground mining and open pits from surface mining and subsequent revegetation of constructed mined soils with grasses and forest trees is the most common approach for long-term restoration. Restoration of mined soils to a stable state depends upon a successful establishment of vegetation. Because soil erosion is a function of slope gradient and length, contouring is a recommended measure to reduce erosion from mined soils with steep topography. Mined soils are typically low in fertility, high in acidity or alkalinity, and prone to compaction. These adverse soil characteristics limit sometimes an optimum growth of plants. Thus, addition of amendments not only boosts plant growth but also improves soil properties by improving nutrient cycling and biological activity.

Table 15.2 Specific methods of mined soil restoration

Vegetative cover	Amendments	Management
• Reforestation • Perennial warm season grasses • Fast growing trees • N-fixing grasses and trees • Forage crops • Meadow • Cool season grasses • Fiber farming	• Animal manure • Biosolids • Fly ash • Green manure • Liming • Fertilizers • Compost • Organic wastes • Sewage and paper sludge	• Backfilling, grading, and leveling topsoil • Terracing and contouring • Livestock management • Drainage systems • Removal of toxic pollutants • Subsoiling • Conservation tillage

15.8.2 Indicators of Soil Restoration

Soil development following restoration is monitored using sensitive indicators such as profile formation, horizonation, erosion rates, change in properties, and accumulation of organic matter. The same soil indicators that are used to evaluate agricultural soils are useful to assess improvement in mined soils. Time, extent, rate, and magnitude of recovery vary with site specific conditions. A properly restored mined soil must have attributes similar to those of pre-mined soils.

15.8.3 Soil Profile Development

Mining operations abolish the original soil horizons and soil profile and alter the overall landscape geomorphology. They truncate the natural soil formation processes and horizonation. For example, at the start of coal surface mining, the top horizons (0- to 50-cm of soil) are removed and stored for reuse. Then, the overburden material, a layer between the topsoil and the coal, is blasted, removed and placed in adjacent areas. The constructed mined soils are a mixture of soil and spoil material and exhibit attributes different from the original soil. Backfilling, spreading, leveling, and grading of topsoil and overburden layers during restoration are undertaken to recreate the original soil profile and landscape form soils with properties and profile characteristics different from adjacent unmined soils. At restoration, soil formation is reset to a time zero, and new processes are set-in-motion to develop new horizons and profile. Restoration of heavily disturbed mined soils is synonymous with (re)creation of soil ecosystems.

15.8.4 Runoff and Soil Erosion

Mined soils are highly unstable and extremely erodible. Even restored sites but with steep slopes and without proper contouring are susceptible to accelerated runoff and soil erosion. Moreover, freshly removed spoils and newly formed mined soils produce low amounts of above- and below-ground biomass insufficient to stabilize the soil surface and reduce erodibility. Soil erosion from steep restored mined soils increases with an increase in slope gradient. On sloping restored mined soils in Spain, sediment loses from small plots were $28.3\,\mathrm{g\,L^{-1}}$ for 30% slope, $31.6\,\mathrm{g\,L^{-1}}$ for 33%, $41.4\,\mathrm{g\,L^{-1}}$ for 46%, and $62.0\,\mathrm{g\,L^{-1}}$ for 56% (Salazar et al., 2002).

Erodibility of reconstructed mined soils differs from that of agricultural soils. Exposed overburden or spoil piles are highly erodible and exhibit gullies due to runoff channeling and steep slopes (Fig. 15.7). In some cases, abundant gravel, stones, and other coarse materials in the spoil reduce sediment loses in contrast with loose topsoil materials. On reconstructed mined soils in western Kentucky, water runoff from plots constructed with mined spoil materials was much higher than that from plots constructed with topsoil only (Mcintosh and Barnhisel, 1993). Runoff and soil erosion from mined soils constructed with the stored topsoil may not differ from those in unmined adjacent sites if slope, vegetation, and management resemble the natural landscapes. In contrast, runoff and sediment yields from abandoned and unrestored mined sites with bare soils can be several orders of magnitude higher than those from restored soils.

Freshly restored mined soils commonly have lower water infiltration rates than unmined soils. Infiltration rates increase, however, with time after restoration due to the increase in macroporosity (e.g., earthworm channels, root biomass). On restored mined soils in Pennsylvania, infiltration rates were only 1 to 2 cm h^{-1} in the first yr of restoration, but after 4 yr, the rates (8 cm h^{-1}) were almost as high as those in pre-mined conditions due to development of macropores (Guebert and Gardner, 2001). Higher infiltration rates result in lower runoff rates and less susceptibility to gullying.

15.8.5 Soil Physical Properties

The removal of topsoil and overburden during mining mix the soil and destroy the natural soil structure and porosity. In addition, increased soil compaction during restoration limits plant growth, reduces soil structure development, and alters soil hydraulic properties. Newly restored soils often have lower aggregate stability, water retention capacity, macroporosity, and water infiltration rates as compared to neighboring unmined sites. The soil physical conditions improve over time following restoration. For example, bulk density in soils restored to forest and pasture decreases with time (Fig. 15.9). Similarly, application of organic amendments reduces bulk density and increases aggregate stability and soil macroporosity. Percent of macroaggregates increased from 24 to 61 whereas that of microaggregates decreased from 57 to 20 following 25 yr of restoration in mined soils in Ohio (Akala and Lal, 2001). The first process during soil recovery is formation of aggregates in response to increases in soil organic matter content, microbial biomass and activity. The second process is the formation of macropores due to the activity of earthworms and proliferation of plant root channels. On mined soils in Ohio, the cumulative infiltration was 53 cm for unmined soils and 6 cm for an unrestored soil while the

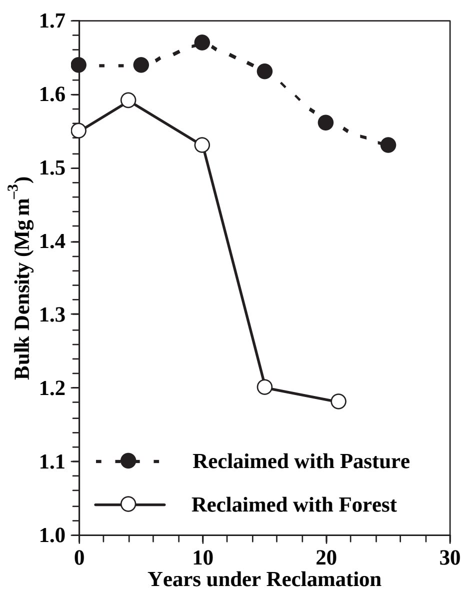

Fig. 15.9 Bulk density of restored mined soils as a function of time (After Akala and Lal, 2001)

mean weight diameter of aggregates was 2 mm for unmined soils and 0.1 mm for an unrestored soil (Shukla et al., 2005).

Summary

Restoration of degraded soils is becoming a necessity for increasing the land area available for crop production. It is essential to recover the soil value and enhance economic development and agricultural productivity. Soil compaction, poor drainage, acidity, sodicity, and salinity are some of the degradative processes. Mining is another major factor that disturbs and degrades soil. Mined soils require specific strategies for their restoration. Reforestation, agroforestry practices, growing perennial trees and native grass species, establishment of cover crops, application of amendments, and establishment of conservation tillage are strategies to restore degraded lands. Conservation tillage can restore partially degraded soils, but severely degraded soils require expensive restoration practices.

Introduction of trees and perennial native grass species is an useful option to stabilize and regenerate the soil fertility. Nitrogen-fixing trees and cropping systems are particularly important to increase soil organic matter and nutrient content. Degraded and marginal lands can be restored through conservation programs (e.g., CRP and CREP in the USA) with the establishment of permanent vegetative cover. Conversion of agriculturally marginal soils to permanent vegetation reduces risks of soil erosion, improves soil properties, promotes wildlife habitat, and enhances biodiversity. Subsoiling, controlled traffic (e.g., reduction in axle loads), reduction of intensive tillage, and mulching are useful strategies to manage compacted soils. Acid soils require liming, fertilization, addition of organic amendments, and use of acid-tolerant crops or plant species for their restoration. Liming is a recommended technique to increase pH and correct acidity.

Saline and sodic soils are restored with installing drainage systems, leaching, subsoiling, organic and chemical amendments, and growing salt-tolerant crop species. The first approach for restoring mined soils includes backfilling, grading, and leveling topsoil. The second step is to establish one or more of the following measures: terracing, contouring, drainage systems, removal of toxic pollutants, conservation tillage followed by reforestation and establishment of fast growing trees, mixture of forage grass, and application animal manure, biosolids, fly ash, green manure, liming, fertilizers, compost, and other amendments.

Study Questions

1. A farmer has applied 45 cm of irrigation water to a field of sweet potatoes that can tolerate salinity up to 3 dS m^{-1} of EC. Calculate the leaching fraction and leaching depth if the EC of the irrigation water is 0.6 dS m^{-1}, and runoff is negligible.
2. During a corn (3 dS m^{-1}) growing season, 10 events of rainfall were recorded. Determine the leaching fraction and leaching depth if the rain depth by event

was: 4.0, 3.6, 3.2, 4.6,4.6, 4.1, 5.0, 4.7, 3.5, and 2.7 cm and EC by event was 0.3, 0.4, 0.5, 0.6, 1.0, 0.8, 0.7, 0.2, 0.9, and 0.1 dS m^{-1}, respectively.

3. A saline soil received a total of 36 cm of irrigation water with an EC of 0.7 dS m^{-1}. Can a salt-sensitive crop with threshold salinity of 2 dS m^{-1} be grown if the crop water requirement is 30 cm.?
4. Calculate the amount of gypsum needed to replace the Na^+ of a saline-sodic soil that contains 10 meq exchangeable Na^+/100 g of soil and 1.2% of sodium carbonate by weight for a total depth of 45 cm. The bulk density of the soil is 1.45 Mg m^{-3}.
5. Estimate the amount of dolomitic limestone required to bring the pH and base saturation of an acid soil from 4.5 and 40% to 7.0 and 90%, respectively. The soil has a CEC of 13 cmol kg^{-1}. The amendment is 90% pure.
6. Discuss the main causes of soil degradation.
7. Describe measures to restore saline-sodic and acid soils.
8. What is the role of conservation in reducing soil salinity and sodicity?
9. Explain how mining operations alter soil profile development and horizonation.
10. Discuss practices used for restoring mined soils.

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Chapter 16
Soil Resilience and Conservation

Soils are prone to degradative processes (e.g., erosion, compaction, salinization, acidification) as discussed in the previous Chapters. Yet, most soils have the inherent capacity to resist exogenous and endogenous perturbations and regain and recover, depending on the severity and duration of the degradative processes, and the intensity of restorative mechanisms. The capacity to recover from perturbations is an important and an inherent attribute of a soil. In other words, a soil possesses an inherent regenerative capacity, which in interaction with proper management, can reverse soil degradation. It is this capacity of the soil to restore itself which forms the topic of this Chapter.

16.1 Concepts of Soil Resilience

Soil resilience refers to the intrinsic ability of a soil to recover from degradation and return to a new equilibrium similar to the antecedent state. Soil resilience can also be defined as the ability of the system to recover its "functional and structural integrity" (Seybold et al., 1999). Functional integrity signifies the capacity of the soil to moderate/improve dynamic functions (e.g., fate and decay of organic compounds, microbial activity, immobilization and transformation of chemicals, provision and recycling of nutrients). Structural integrity signifies the intrinsic capacity of the soil to improve its structural properties itself (e. g., soil aggregation, porosity) and return to the initial conditions.

The term soil resilience has evolved from the theory of ecological resilience widely used to describe reactions of terrestrial and aquatic ecosystems to anthropogenic and natural perturbations. An accurate definition of resilience is difficult because of the dynamic, variable, and heterogeneous nature of the soil system (Table 16.1). Not all soils respond in a similar manner to applied stresses. Thus, the concept of soil resilience can be as complex as the soil system. An operational definition of soil resilience and methods of its assessment is the topic of an intense debate. Resilience is specific to each soil and depends on the interaction of soil physical, chemical, and biological processes. Thus, a single definition of soil resilience may not fully express the variable and complex behavior of soils. An

Table 16.1 Some definitions of resilience

Basic Definitions
[1]The capability of a strained body to recover its size and shape after deformation caused especially by compressive stress. An ability to recover from or adjust easily to misfortune or change.
Engineering resilience
[2]Time of return to a global equilibrium following a disturbance.
Ecological resilience
[3]The amount of disturbance that a system can absorb before it changes to an alternative stable state.
Ecosystem resilience
[4]The magnitude of disturbance that a system can absorb or accommodate before it changes its structure by changing the variables and processes that control its behavior.
Soil Resilience
[5]The capacity of a soil to recover its structural and functional integrity after a disturbance.
[6]The ability of a soil ecosystem to return to dynamic equilibrium after disturbance.
[7]The ability of the soil to resist or recover from an anthropogenic or natural perturbation.
[8]The ability of the system to revert to its original or near original level performance or state that existed before the impressed forces altered it.
[9]The capacity of a soil to resist change caused by a disturbance.
[10]Processes that enable soils to counteract stress and alterations.

[1]Merriam-Webster Online Dictionary (2006), [2,3]Dorren and Imeson (2005), [4]Holling and Meffe (1996), [5]Seybold et al. (1999), [6]Blum and Aguilar (1997), [7]Lal (1997), [8]Eswaran (1994), [9]Rozanov (1994), and [10]Szabolcs (1994).

appropriate and a comprehensive definition must be based on soil's ability to recover from perturbation to perform a specific process or function.

Discussions of soil resilience in the literature tend to stress "return to its original state" or "initial condition" following perturbation. Soil resilience does not necessarily mean that the system will bounce back to a state identical to that prior to perturbation. What it means is that the perturbed soil system recovers to a state where its performance is not significantly different from the one before. Analogies between soil resilience and a common spring which implies that a soil would return to its identical original position following perturbation may not apply to most soils.

16.2 Importance

The theory of soil resilience allows the understanding of soil functioning in relation to soil stability and productivity. Surprisingly, soil resilience has neither been addressed nor defined in as much detail as it deserves. Resilience is a key soil attribute in that it stands for the capacity of the soil to recover from continuous and persistent anthropogenic stresses. If it had not been for this vital soil attribute, all managed soils would have ceased to produce ecosystem services long ago.

The concept of soil resilience is gaining importance in the context of increased risks of soil degradation and growing concerns about the climate change. How will a soil respond to change in climate? This is an unanswered question that necessitates an urgent consideration. The climate-induced changes in soil temperature, rainfall amount, and drought would directly affect soil structural resilience. While some studies have suggested that moderate increases in soil temperature would enhance plant growth and biological activity, thereby increasing soil organic matter, extreme events such as intense rain storms, flooding and droughts may severely diminish the ability of a soil to recover. Experimental data on soil resilience from long-term studies simulating changes in global climate are needed for principal soils to understand the magnitude and direction of effects. It is crucial to understand how soil responds to climatic fluctuations, and the nature of factors and processes that control this response. It is likely that recurrent degradation events induced by climate change may cause loss of soil's resilience by altering both biotic conditions and soil processes.

16.3 Classification of Soil Resilience

Soils are grouped into different categories based on their degree of resilience (Table 16.2). The classification is often based on the most dominant degradation factor. In some soils, water erosion can be more damaging than wind erosion. In others, drainage and salinization can be major degradation factors. Highly resilient soils can recover rapidly after degradation or even resist degradation stress because of favorable extrinsic (e.g. geomorphology, climate) and intrinsic properties (e.g., profile depth, organic matter content). Conversely, non-resilient soils would collapse rapidly under degradative stress and fail to recover even under favorable conditions. Rate and magnitude of soil response or recovery depend on the specific degradation factors. For example, severely eroded soils over extended time periods may never recover. In comparison, slightly or moderately eroded soils may be relatively resilient and recover rapidly. Differing responses of soils to degradation processes are confounded by the complex nature of each soil. Even under the same management, some soils can regain their pre-disturbance status sooner than others because of differences in profile depth, horizon thickness, and soil organic matter pool. A major difficulty for classifying soil resilience is the identification of parameters that enable comparisons of resilience within, between, and among soils. Available classifications of resilience are qualitative and have limited applicability unless made quantitative based on solid parameters of evaluation (Table 16.2).

Two essential components of soil resilience are amplitude and elasticity (Benitez et al., 2004). Amplitude refers to the time it takes to recover to the initial condition after disturbance, while elasticity refers to the speed of recovery after the application of stress has ceased. How fast a soil recovers from disturbance is crucial to identify and adopt proper land use and management strategies in relation to the desired productivity. The projected time for the recovery of soil properties after degradation depends mainly on climatic, ecologic, and management conditions.

Table 16.2 Classification of soil resilience for soil erosion [After Lal (1997) and Seybold et al. (1999)]

Class	Resilience	Description	Response to soil erosion	Soil characteristics
0	Highly resilient	• Rapid recovery, high buffering	• Highly resistant or non-erodible	• Very deep and very high organic matter content, aggregate stability, and water infiltration rates with profile depth
1	Resilient	• Recovery with improved management	• Resistant or very slightly erodible	• Deep and high organic matter content, aggregate stability, and water infiltration rates with profile depth
2	Moderately resilient	• Slow recovery with high input	• Moderately resistant or erodible	• Moderately deep and moderately high organic matter content, aggregate stability, and water infiltration rates.
3	Slightly resilient	• Slow recovery even with change in land use	• Low resistance and highly erodible	• Shallow soils and low organic matter content, aggregate stability, and water infiltration rates
4	Non-resilient	• No recovery even with change in land use	• Non-resistant or extremely erodible	• Very shallow and very low organic matter content, aggregate stability, and water infiltration rates

16.4 Soil Disturbance

Soil disturbance refers to abrupt or gradual changes which alter soil processes and properties, and the normal functioning of the soil system. A short summary of the main types of soil disturbance is presented in Table 16.3. There are natural and anthropogenic disturbances. Disturbances are part of the soil ecosystem, occur at all times, and are often necessary to perform essential management operations for producing the needed goods and services. Indiscriminate soil disturbance, referred to as soil degradation, however, leads to major changes in physical, hydrological, chemical, and biological processes, affecting soil function. Agriculture is one of the greatest anthropogenic activities that cause soil degradation. Unlike anthropogenic disturbances, natural disturbances are not preventable. Mismanagement of soils with intensive tillage and monocultures create stresses in the system, causing rapid and non-reversible changes in the soil. Urban sprawl is a growing contemporary factor that alters the natural landscape conditions. Episodic events such as drought and flooding often trigger soil degradation in managed ecosystems.

Table 16.3 Types of degradation factors affecting soil resilience

Natural disturbance	Anthropogenic disturbance
• Landslides • Earthquakes • Fires • Wind storms • Rainstorms (e.g., runoff, raindrop impacts) • Drought • Floods • Water table fluctuations	• Deforestation • Tillage (e.g., moldboard plowing) • Farming practices (e.g., up and down slope tillage) • Cropping systems (e.g., monocultures) • Fertilization (e.g., inorganic fertilizers) • Pesticide application • Irrigation with poor quality water • Salinity caused by management • Traffic (e.g. animal, equipment traffic) • Grazing • Urban development (e.g., scraping, excavation, and creation of impervious surface)

16.5 What Attributes Make a Soil Resilient?: Factors

Soil resilience depends on the pre-disturbance conditions of the system. Soils that are well-structured, deep, and have high soil organic matter content exhibit high resilience. The combination of intrinsic textural and structural properties controls the soil resilience. Surface cover improves soil resilience against erosion. Dense cover of vegetative canopy and residue on the soil surface is a critical companion to maintain/increase the soil resilience.

Soil resilience is affected by the same factors that also govern soil formation (Fig. 16.1). Factors of soil resilience refer to the biophysical parameters including parent material, soil intrinsic properties, soil geomorphology, vegetation, and climate, which interact and revolve over time (Table 16.4). One factor could be more influential than the other, depending on the soil type. As a result, the dominant factor determines sequences in soil formation and affects the rate and magnitude of soil recovery. Factors and processes that affect soil resilience are continuous, simultaneous, and interdependent. In addition to the five-soil forming factors, external mechanisms such as the socio-economic characteristics of farmers/landowners and land policy programs influence the soil resilience. Understanding the cause-effect relationship of soil resilience is critical to the long-term soil productivity and development of proper land use and management strategies to improve soil functions.

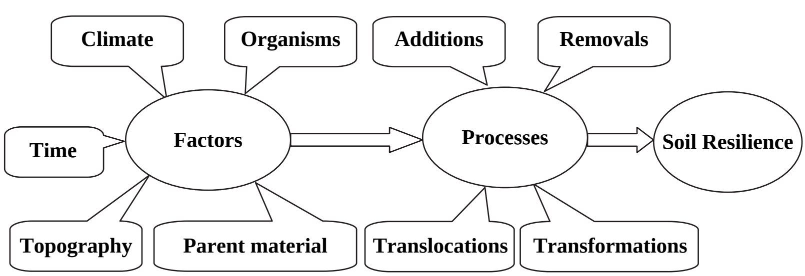

Fig. 16.1 Factors and processes of soil formation affecting soil resilience

Table 16.4 Factors affecting soil resilience (After Lal, 1997)

Soil characteristics	Landscape characteristics and biota	Climate and time
• Parent material (residual, colluvial, alluvial, marine, lacustrine, glacial, eolian, and organic) • Soil physical properties (e.g., texture) • Soil chemical properties • Soil biological properties (e.g., microbial biomass)	• Land topography (aspect, gradient, length, and shape) • Soil biota (flora and fauna) • Land use and management • Level of landscape disturbance	• Precipitation (e.g., rainfall, snow) • Temperature (maximum and minimum) • Radiation • Relative humidity • Length of soil degradation and restoration • Rate of soil formation

16.5.1 Parent Material

The nature of parent material determines the soil texture, which in turn influences the soil resilience because it governs the fluxes of water, air, and heat through the soil. For example, soil resilience mechanisms such as leaching of chemicals and translocation of fine soil particles within the profile (eluviation/iluviation) are controlled by soil textural differences. Total soil pore space increases with increase in clay content. Thus, clayey soils retain more water than sandy soils. Sandy soils are, however, more permeable than clayey soils as they have large and well-connected pores. Rates of runoff and soil erosion are also functions of soil texture. Soils with high silt and fine sand contents are more susceptible to rill, interrill, and gully erosion than either clayey or sandy soils. This does not, however, imply that sandy and clayey soils are more resilient than silty soils. The sustainable use depends on the management. Degraded silty soils of loess or alluvial origin can be more rapidly revegetated or restored if they have thick horizons and deep profile. The clay content and mineralogy are also crucial components which influence soil resilience. Soils containing shrink-swell or high activity clays (e.g., montmorillonite) are more resilient to compaction than those containing predominantly low activity clays and low shrink-swell capacity (Seybold et al., 1999). Clays that disperse easily in water are highly erodible and less resilient. Aggregate stability, crusting, surface sealing, and water transmission characteristics related to soil erodibility are directly influenced by water dispersion properties of clays. The physical and chemical composition of primary and secondary particles also influences the macro-scale physical and chemical properties of the system.

16.5.2 Climate

Climatic parameters influence the magnitude of soil resilience. The capacity of a soil to recover from a disturbance is lower under drier than wetter climates (Lal, 1994). Climatic factors affecting resilience include precipitation, temperature, radiation (albedo), air humidity, and evaporation demand. Fluctuations or seasonal distribution of climatic parameters have a major effect on soil biota and the intensity of weathering of the parent material. Effective precipitation and temperature are drivers of all physical, chemical, and biological processes of soil resilience. Rain water that infiltrates into the soil carries fine soil particles and dissolved substances and contributes to soil formation and restoration. Therefore, in regions with low annual precipitation, most of the processes responsible for the differentiation of horizons are less intense because of the absence of percolating water. Temperature is essential to moderate plant growth and microbial processes, which are sensitive indicators of soil resilience. Rates of soil biochemical reactions double for every 10°C increase in soil temperature (Brady and Weil, 2001). Climate is the most important soil factor that sets the biotic activities (e.g., plant growth, soil animals). Humid and temperate climates promote rapid growth of plants in degraded soils as compared to semi-arid and arid climates with slow soil recovery.

16.5.3 Biota

16.5.3.1 Flora

Soils are more resilient under vegetative cover than when denuded. Soil erosion and physical degradation set in when protective vegetal cover is removed. Increase in vegetative cover decreases soil erosion by protecting the soil against raindrop impacts and reducing soil splash and detachment. Plant roots create a network of channels through the soils. Fibrous grass roots are often concentrated near the soil surface and form continuous and fine pores while adding the vital soil organic matter. Roots of trees and shrubs penetrate deep into the soil profile influencing water infiltration and nutrient cycling with depth and integrating the surface and subsoil processes. The interactive action of the above-ground surface cover and the below-ground biomass (e.g., plant roots and residues) contributes to enhancing soil resilience. Vegetation imparts significant differences in soil properties essential to improving soil resilience. For example, forest soils often have lower bulk and particle densities compared to agricultural land use for the same soil. The accumulation of soil organic matter in grasslands generally results from the addition of below-ground biomass (root system) whereas that in forest soils results from the above-ground biomass (surface litter or leaves falling on the soil surface).

16.5.3.2 Fauna

A good soil is teeming with life, and is home to an extraordinary number of macro- and micro-organisms, which are the drivers of key processes of soil resilience. Soil animals can be divided into three main groups: microfauna ($<100\,\mu$m diameter), mesofauna (100 μm to 2 mm diameter), and macrofauna (>2 mm diameter) (Bradford et al., 2002). These animals may differ in size and activity but their essentiality to soil resilience is similar. Soil fauna are key determinants of soil resilience because they affect nutrient turnover and cycling, soil organic matter turnover, and soil aggregation. Changes in population and composition of soil animals impact the soil functioning. The macrofauna (e.g., earthworms, termites, ants) through pedoturbation, mixing of soil materials, contribute to structuring of the soil and thus are called by many as "ecosystem engineers" (Wright and Jones, 2004). Earthworms mix inorganic and organic soil particles and improve porosity and aggregation. They also create inter-connected, heterogeneous, vertical, and extensive biological macropores (e.g., wormholes). These biopores become the pathways for rain and runoff water infiltration, decreasing the runoff rate and the volume. The meso- and micro-fauna are no less important to soil dynamics. They affect chemical reactions (solubilization and transformation) and exchanges between plant roots and soil. Interactions among earthworms, microbes, and plant roots influence the soil restoration through soil structuring, nutrient cycling, and the turnover of soil organic matter.

16.5.3.3 Anthropogenic Influence

Anthropogenic activities are the leading cause for altering soil formation and degrading well-formed soils. Deforestation, land clearing, intensive plow tillage, irrigation, overgrazing, and mining are among the major degradative interventions and perturbations of humans. Mechanized tillage overturns, mixes, and exposes the soil to climate elements. As a result, the soil is exposed to a range of degradative processes. What is required is drastic changes in soil management and social and political predisposition to counteract soil degradation and enhance soil resilience. The socio-economic and political forces affecting soil resilience include landowner's predisposition, land policies, incentives, and education.

16.5.4 Topography

Differences in elevation, slope length, slope steepness, and landscape position influence the soil restoration. The steeper is the slope gradient of degraded soil, the more difficult it is to restore/rehabilitate it. Sloping soils are susceptible to erosivity of concentrated runoff and are prone to accelerated soil erosion as the water infiltration opportunity time is drastically reduced. Topography also influences the transport and accumulation of soluble salts. In some arid and semiarid regions, low lying areas often accumulate salts, soluble toxic and non-toxic chemicals because of flooding and poor drainage. Growth and diversity of plants interact with soil topography. Sloping soils grow sparse vegetation and have reduced plant diversity because of limited water storage, poor horizonation, and shallow profile as compared to deep soils in flat terrains. Deep-rooted species (e.g., trees) are often confined to footslope positions leaving upland landscape positions degraded and with sparse vegetation. Lower landscape positions may be more resilient because of high soil organic matter content and sediment accumulation. Lower landscape positions also have lower bulk density and higher volumetric water content than upper landscape positions.

16.5.5 Time

Soil restoration generally follows a sigmoidal response and is a function of duration for which the restorative practices have been adopted. When soil degradation ceases, the curve of soil recovery rises as time passes. It often takes a long period of time before the soil returns to a desired level of performance or functionality. Lal (1994) observed that when a degraded soil was reverted to natural fallow, the infiltration rates increased 12-fold in one yr, 22-fold in two yr, 60-fold in three yr, and 100-fold in four yr (Fig. 16.2). When other factors are favorable, soil organic C also accumulates with time following degradation until it reaches a dynamic equilibrium level. Organic inputs darken the upper surface of the soil and contribute to the differentiation of A horizon within 10 or 20 yr. Development of B horizon requires, however, longer periods of time (>100 yr). Formation of 0.3–1 cm of new soil requires about 100 yr in most soils. The speed of soil recovery depends on climate. In regions with abundant rain, translocations and transformations of organic and inorganic materials occur more rapidly than in dry regions, modifying the soil profile characteristics. Tilled soils exhibit no differences in upper surface horizons or biological activities. Thus, the clock resets for these systems when degradation ceases.

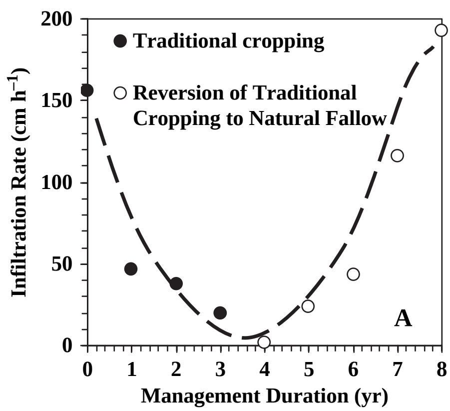

Fig. 16.2 (**A**) Water infiltration as a function of type of management and (**B**) soil C buildup with time after degradation (After Lal, 1994)

16.6 Soil Processes and Resilience

Processes affecting the magnitude and rate of soil recovery include physical, chemical, and biological mechanisms (Table 16.5). These mechanisms interact and occur simultaneously, influencing various interrelated soil functions. Rate and magnitude of soil processes vary among themselves and affect the speed and time of soil recovery. Processes such as decomposition of soil organic matter and activity of microorganisms are more dynamic and rapid than weathering of minerals. Biological processes are responsible for nutrient cycling, absorption and release of nutrients, and processing of organic and inorganic components. Thus, soil resilience may

Table 16.5 Processes affecting soil resilience (After Lal, 1997)

Physical	Chemical	Biological
• Physical weathering (e.g., freezing or thawing, exfoliation, crystallization) • Soil water, air, and heat fluxes • Macro-and micro-aggregation • Flocculation • Shrinking and swelling • Clay formation	• Weathering (e.g., cation, solution, hydration, hydrolysis, oxidation) • Immobilization • Transformation • Nutrient cycling • Buffering capacity • C sequestration (e.g. formation of organomineral complexes)	• Biological weathering (e.g. burrowing) • Root growth • Activity of macro- and micro-organisms • Soil organic matter decomposition and accumulation • Biodegradation and biotransformation

mainly apply to biological processes which are reversible, sensitive, and rapidly affected by soil management unlike slow processes such as weathering of rocks or parental material. The major processes of soil resilience are related to those of soil formation (Table 16.5).

Addition of eroded sediments and dust from neighboring environments and translocations of materials by water and soil organisms to deeper horizons are essential physical mechanisms of soil formation and restoration (Fig. 16.3). Soil aggregation is a fundamental process to soil erosion reduction. Immobilization and transformation of nutrients are part of nutrient cycling. First, organic materials react with clay particles through adsorption. Second, clay surfaces polymerize humic substances. Third, polymerized organic compounds are physically and chemically sequestered by clay crystals inaccessible to soil organisms. Interactions among polysaccharides, humic and fulvic acids, and clay particles moderate soil aggregation and soil organic matter dynamics (Tisdall and Oades, 1982). The chemical

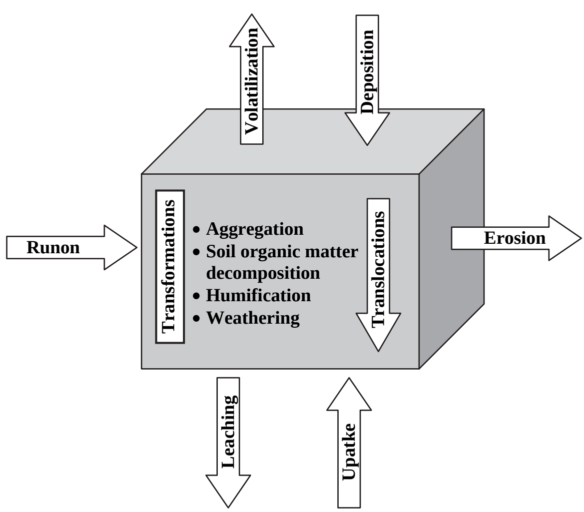

Fig. 16.3 Processes affecting soil formation and resilience (After Buol et al., 1989)

processes operating along with physical and biological processes are crucial drivers of soil recovery. Soil microorganisms, plants, and animals contribute to soil resilience by weathering minerals, creating pathways for the translocation of materials, and recycling nutrients. The biological mechanisms occur simultaneously with physical and chemical mechanisms.

16.7 Soil Erosion and Resilience

Susceptibility of a soil to water and wind erosion is a function of physical, mechanical, chemical, and biological resilience at different spatial and temporal scales. A resilient soil can withstand soil erosion and regenerate itself to a stable condition upon changes in degradative processes. Resilience of the soil is highly dynamic, somewhat uncertain, and varies greatly during the recovery stages. The rate of recovery may follow a sigmoid path. Runoff and soil erosion rates are often lower in high resilient soils attributed to the higher water flux through the soil.

The most direct effect of soil erosion is the loss of topsoil, which is being eroded at a rate of about 7% worldwide (Papendick, 1994). On sloping environments, the entire or large fractions of the A horizon can be removed by erosion, bringing to the surface the fragile horizons of low fertility and poor structural properties unsuitable for crop production. The uppermost few centimeters of the soil bears the highest amount of essential nutrients and the most suitable physical attributes for plant growth. Accelerated erosion can remove as much as one or two centimeters of topsoil in a few hours or days, but it could take hundreds of years to naturally recover or form such a thin layer of soil. Lost topsoil may be regenerated by weathering but this is an extremely slow process. The rate of soil formation ranges between 0.01 and 0.003 cm yr^{-1} in most tropical soils. Soil resilience is often measured by the rate of topsoil development after severe erosion. A soil can be renewable if only the rate of soil depletion is lower or equal to the rate of soil formation. Recovery of topsoil after disturbance requires a long-term monitoring of physical, chemical, and biological attributes to assess the amplitude and elasticity of soil resilience.

16.8 Soil Resilience and Erodibility

16.8.1 Soil Physical Properties

Dynamic parameters affecting the soil erodibility are directly affected by the degree of soil resilience. Soils of high resilience are characterized by better soil structure (e.g. higher macroporosity) and water transmission characteristics (e.g., infiltration capacity). Highly resilient soils have lower bulk density and higher percentage of water stable aggregates, plant available water, and water retention than those of low resilience (Table 16.6). Some properties can regain their original values much quicker than others owing to differences in their intrinsic dynamic attributes. On an

Table 16.6 Changes in surface (<20 cm depth) soil physical properties during recovery

Soil	Practices promoting resilience	Soil properties	Resilience	
			Before	After
Sandy clay loam[1]	2-yr natural fallow (grasses and native legumes)	Water-stable aggregates (%)	44	66
		Infiltration rate (cm h^{-1})	13	32
Sandy clay loam[2]	5-yr under Kallar grass	Saturated hydraulic conductivity (mm d^{-1})	0.04	56
		Bulk density (Mg m^{-3})	1.7	1.5
		Porosity (mm^3 mm^{-3})	0.4	0.4
		Available water (kg kg^{-1})	0.1	0.2
Sandy loam[3]	6-yr natural fallow	Bulk density (Mg m^{-3})	1.4	1.2
		Penetration resistance (kPa)	295	132

[1]Chirwa et al. (2004), [2]Akhter et al. (2004), and [3]Tian et al. (2000).

Udic Ustochrept severely affected by landslides, soil macroporosity of the topsoil had recovered its original value in 24 yr, bulk density 90% in 50 yr, particle density 71% in 79 yr, aggregate stability 33% in 10 yr and 100% in 45 yr (Sparling et al., 2003). Soil dynamic properties including bulk density, porosity, saturated hydraulic conductivity, water infiltration, water-stable aggregates, and penetration resistance are significantly improved when degraded soils are reverted back to natural fallow (Table 16.6).

Intensive traditional cropping results in a rapid soil degradation, altering soil hydrological properties, but when the traditional system is reverted to natural fallow, the soil properties gradually improve following a sigmoidal function. Continuous cultivation reduces soil structural stability and that reversion to natural fallow increases macroaggregation, which is crucial to reducing soil erosion. The breakdown of macro-aggregates in cultivated soils increases soil erosion linearly while improvement in macro-aggregation during recovery also decreases the soil erosion accordingly.

Structural resilience is the soil's ability to recover its structural properties in response to the introduction of a restorative land use and management. Any mechanical operation disturbs first the soil structural properties and can eventually lead to soil degradation. Structural resilience is a fundamental soil attribute that controls fluxes of water and air through the soil system. Discussions about soil resilience must also address soil structural resilience because most of the dynamic soil properties depend on the nature and resilience of soil structure. Soil structure is a dynamic property and is the framework that moderates and integrates all the soil processes influencing soil resistance to erosion and mechanical stresses. Soils that have the ability to recover their structural properties following degradation are suited to agricultural land use and are valuable asset.

16.8.2 Soil Chemical and Biological Properties

Resilient soils not only improve their physical properties rapidly but also enhance chemical and biological properties. Total C and N concentrations, microbial processes, biochemical properties, and soil structural parameters (e.g. bulk density, porosity) have the highest resilience while properties related to solid phase (e.g., particle density) have the lowest. Changes in total C and N concentrations and microbial biomass are often used as sensitive parameters to monitor the soil recovery (Sparling et al., 2003). Soil micro-organisms, through their dynamic activities, play a major role in moderating soil organic matter decomposition and enrichment. Microbial activities act upon the above- and below-ground plant biomass and contribute to soil fertility concentrations. Organic C and N recover rapidly during initial periods ($<$15 yr) but it could take 100–150 yr to reach the pre-undisturbed or equilibrium levels. Fifty nine yr following degradation, microbial C, pH, total C and N concentrations, mineralizable N, total P, and CEC had recovered 71–85% while soil respiration, invertase and sulphatase activities had recovered to 94–110% as compared with non-degraded soils (Sparling et al., 2003). The fast turnover rates of microbial biomass help restore soil structural and water transmission properties in degraded soil ecosystems. Total C and N concentrations can be used as sensitive measures of soil resilience in absence of data on biochemical properties.

16.9 Soil Resilience and Chemical Contamination

Soil contamination with either solid or liquid toxic substances from spills is a major environmental concern (USEPA, 2006). Water and wind carry chemicals from contaminated areas and deposit them in the soil. Two main treatments of contaminated soils include: (1) treatment (e.g. flushing, incineration) of the affected soil through extraction or in-situ conditions, and (2) containment of soil in place or isolation (e.g., plastic cover to prevent the contaminants from spreading by rain, runoff, and traffic). The second approach of treating contaminated soils relies partly on the intrinsic capacity of the soil to recover itself. Resilient soils through their inorganic and organic compounds and macro- and micro-organisms can degrade or transform toxic chemicals over time.

Soils have the ability to buffer successive inputs of agricultural chemicals and other pollutants causing soil and water pollution. Amplitude and elasticity are a function of the period and magnitude of exposure of the soil to contaminants. For example, soil dehydrogenase activity ceases when the olive-mill solid waste is applied, but, after a few months, it recovers and returns to the original levels (Benitez et al., 2004). The capacity of a soil to degrade chemicals is specific to soil type and management. Well-developed soils with high organic matter content (e.g., Mollisols) can be more resilient than less developed soils (e.g., Entisols). Moreover, intensively cultivated soils often have less resilience than non-agricultural soils with other factors remaining the same. Interaction of soil physical and biological processes is responsible for the soil resilience in metal-contaminated environments.

16.10 Indicators of Soil Resilience

Changes in soil physical, chemical, and biological properties are used as indicators of soil resilience (Table 16.7). These properties must be monitored from the onset to the end of the restoration period to determine the rate and magnitude of recovery. Changes in some soil characteristics such as horizonation and profile depth are slow and may take decades before significant changes occur. On the contrary, changes in dynamic properties (e.g., aggregation, microbial activity) are often detectable shortly after adoption of restorative practices. An ultimate indicator of soil resilience is the return of the soil to a state where it performs a specific function (e.g., crop production) in a way that is not significantly different from that prior to being degraded. Soil aggregation is an early indicator of soil resilience and is quantified by measuring specific aggregate properties such as stability, size distribution, strength, and porosity (Cammeraat and Imeson, 1998). Accumulation of soil organic matter is another process which affects not only soil fertility but also aggregation, water retention, and biological activity. The ideal approach to monitor soil resilience is the use of long-term and replicated experiments where soil properties are measured and compared with a valid baseline. Standard sensitive indicators for assessing soil resilience must be developed. The challenge is to define a small but complete set of sensitive properties enough to detect significant differences in soil function.

Table 16.7 Soil intrinsic properties whose changes are used as indicators of soil resilience

Physical properties	Chemical properties	Biological properties
• Horizonation	• Soil pH	• Microbial biomass
• Color and depth	• Essential nutrients	• Microbial activity
• Clay content	• Total organic C	• Earthworm population and activity
• Bulk density and porosity	• Total N content	• Above- and below-ground biomass
• Water-stable aggregates	• C:N ratio	• Rooting depth
• Soil temperature and air permeability	• Particulate organic matter	• Root abundance
• Water infiltration	• Particulate P	• Biodiversity
• Water retention capacity	• Cation exchange capacity	• Plant growth and population
• Cone index and tensile strength	• Oxidation and reduction	
• Coefficient of linear extensibility and plastic limit	• Sodicity	

16.11 Measurements of Resilience

Soil resilience can be measured by assessing the rate and magnitude of recovery of soil properties (Seybold et al., 1999). One of the approaches consists in monitoring directly the rate of soil recovery using long-term field experiments. Through this approach, rates and maximum levels of recovery are evaluated based on specific soil functions. Soil productivity, topsoil development, and soil structuring are examples of specific soil functions. A related approach consists in measuring selected indicators of soil resilience (Table 16.7).

16.12 Modeling

Several models and mathematical functions have been developed to simulate soil resilience. Modeling is essential to determine the length of time that it would take for a soil to reach the equilibrium state or regain its potential to perform ecosystem services.

16.12.1 Single Property Model

Herrick and Wanderm (1998) determined that the soil recovery rate (R_r) as

$$R_r = \frac{d\left[\dfrac{(B-C)}{(A-C)}\right]}{dt} \tag{16.1}$$

where A is predisturbance level, B is stabilized equilibrium level, and C is level immediately before the start of restoration. Equation (16.1) shows that soil resilience is the average rate of change of soil quality with respect to time (t). The time interval within which a change in soil quality occurs is critical for computing soil resilience.

Example 1. Estimate the recovery rate for a soil which reached a 70% of dynamic equilibrium level after being degraded to 20% of the original level.

$$R_r = \frac{(B-C)}{(A-C)} = \frac{70-20}{100-20} = 0.625 = 62.5\%$$

Soil recovery levels are often lower than the original level, depending on the severity of disturbance or damage and restoration length.

16.12.2 Multiple Property Models

Ellenberg (1972) suggested that the stress capacity (SC) of an ecosystem is as shown in Eq. (16.2)

$$SC = \frac{(100 - D \times L) \times R}{10} \tag{16.2}$$

where D is disposition, the ease with which a disturbance reaches a system, L is system's susceptibility to disturbance, and R is the restoration of the system. The three factors are rated on a scale from 0 to 10.

DeAngelis (1980) proposed that the return time (T_r) to equilibrium following a disturbance at t = 0 is.

$$T_r = \int_0^{\infty} dt \frac{\sum_{i-1}^{3} \left\{ \left(X_i\,(t) - X_i^* \right)^2 / X_i^{*2} \right\}}{\sum_{i-1}^{3} \left\{ \left(X_{1.i} - X_i^* \right)^2 / X_i^{*2} \right\}} \tag{16.3}$$

where X_i^* is the equilibrium value of the ith component of the model, $X_{1.i}$ is the initial displacement of the ith component, and $X_i\,(t)$ is the instantaneous value following the disturbance.

Lal (1994) proposed the following models to predict the renewal rate of a soil system following degradation

$$S_r = \frac{-dS_q}{dt} \tag{16.4}$$

and

$$S_r = S_a + \int_0^t (S_n - S_d + I_m)dt \tag{16.5}$$

where S_r is soil resilience, S_q is soil quality, t is time, S_a is the initial condition, S_n is rate of soil renewal, S_d is the rate of soil degradation, and I_m is management input.

The relative recovery rate (R_r) of a soil property is the change (decrease/increase) with respect to an antecedent value

$$R_r = \left(\frac{Initial - New}{Initial} \right) 100$$

Szabolcs (1994) presented a model to predict soil function based on soil resilience and resistance as

$$S_r = BC_{ph} + BC_{ch} + BC_b + \int \frac{dPSF}{dt} + \int \frac{dAF}{dt} \tag{16.6}$$

where S_r refers to soil potential to resist changes and recover soil functions, BC_{ph} is the physical buffering capacity, BC_{ch} is the chemical buffering capacity, BC_b is biological buffering capacity, *PSF* is pedological soil fluxes, and *AF* is anthropological soil fluxes.

Rozanov (1994) proposed a model to predict soil resilience based on the principle of physics of a common spring. Soils, similar to a spring, would return to their original condition once the stress or pressure is released as

$$\frac{dA}{dx} = -kx \tag{16.7}$$

where A is the amount of stress or disturbance needed to alter soil quality, x is change in soil property, k is resilience coefficient and is specific to each soil, land use, and management system.

Kay et al. (1994) proposed that the recovery of soil structural characteristics (S) is a function of time (t) and is given as

$$S = f(t) \tag{16.8}$$

The S in Eq. (16.8) depends on the three functions including land use and management, biological, and weather characteristics. This model can be used for projecting changes in soil structural processes leading to either degradation or recovery. The analytical form of Eq. (16.8) for resilience of structural properties is

$$S_{(t)} = S^0 + \Delta S_{max}[(l - e^{-k(t-t_1)})] \tag{16.9}$$

where S^0 is the value of S at the start of the new land use when t = 0, ΔS_{max} is the maximum projected change in S, k is the rate constant, and t_1 is the time between the introduction of the new land use and the time of measurement ($t \neq 0$). The ΔS_{max} is the resilience potential if the degradative process is minimized. Resilience potential is maximum recovery in structural characteristic that a soil can undergo under the reduction or elimination of the degradative processes (Kay et al., 1994). The exponential function indicates that the magnitude of changes in soil structural properties is large in early stages and it decreases with increase in time.

Example 2. Estimate the infiltration rates over 1, 5, 15, and 30 yr following the establishment of perennial vegetation on a previously degraded soil. The infiltration rate at the time of establishment of the new land use was 15 cm h^{-1} and the maximum projected change in infiltration rate is 38 cm h^{-1}. Assume k equal to 0.55 and $t_1 = 0$.

$$S_{(t)} = S^0 + \Delta S_{max}[(l - exp^{-k(t-t_1)})] \tag{16.10}$$

$$S_{(t)} = 15 + 38[(l - exp^{-0.55t})] \tag{16.11}$$

Year 1: $S_{(t)} = 15 + 38[(l - exp^{-0.55\times1})] = 31.07\,\text{cm}\,\text{h}^{-1}$

Year 5: $S_{(t)} = 15 + 38[(l - exp^{-0.55\times5})] = 50.57\,\text{cm}\,\text{h}^{-1}$

Year 15: $S_{(t)} = 52.99\,\text{cm}\,\text{h}^{-1}$

Year 30: $S_{(t)} = 53\,\text{cm}\,\text{h}^{-1}$

Singh et al. (2001) developed simplified equations to predict the recovery time of dynamic soil properties in hilly and degraded forest ecosystems as

$$OC = 12.496t^{\,0.191} \tag{16.12}$$

$$TN = 1.266t^{\,0.171} \tag{16.13}$$

$$MB - C = 130.36t^{\,0.398} \tag{16.14}$$

$$MB - N = 13.377t^{\,0.348} \tag{16.15}$$

where *OC* is organic C (Mg ha^{-1}), *TN* is total N (Mg ha^{-1}), *MB-C* is microbial biomass C (μg g^{-1}), *MB-N* is microbial biomass (μg g^{-1}), and t is the time of recovery (yr).

Sparling et al. (2003) modeled the rate of topsoil development (*TR*) affected by slippage using

$$TR = A_i(1 - exp(-k_i t) \tag{16.16}$$

and that accounting for changes in soil structural characteristics during recovery

$$TR = A_i - (A_i - A0_i)exp(-k_i t) \tag{16.17}$$

where t is the landslip-age, i is sampling time, and k is the constant that controls the speed at which the recovery curve rises or falls. According to Sparling et al. (2003) the increasing recovery curve passes through zero when $t = 0$, and rises to an asymptote A_i while the decreasing curve passes through $A0$ when $t = 0$ and falls to the asymptote A.

16.13 Management Strategies to Promote Soil Resilience

Soils can be able to develop a self-regenerating system against degradative processes thorough adoption of restorative management systems. Practices leading to soil degradation should be systematically matched with practices leading to improvement in soil resilience. The key to improving the resilience of soils is the adoption of practices that increase the input of soil organic matter. Organic matter improves the soil pore structure, increases water infiltration, and reduces soil compaction and runoff and soil erosion. Improvements in microporosity and pore structure are essential to water retention and transmission properties of the soil. High quantities of soil organic matter act like a sponge, lowering the compressibility of the soil but enhancing resilience upon release of stresses. Restoration of degraded soils requires the transformation in farming practices, land use, and human attitude. Practices used for soil restoration are discussed in Chapter 15.

One of the strategies to improve resilience of moderately degraded soils is adoption of conservation tillage (Table 16.8). This shift in management sets in motion the

Table 16.8 Case studies of restoration management strategies to improve soil resilience

Region	Management strategy
	Natural fallow
Bangladesh[1]	Establishment of natural fallow decreased soil erosion from 18 to 3 Mg ha^{-1} yr^{-1} to levels similar before the adoption of intensive cultivation.
Northeastern Mexico[2]	Water-stable macroaggregates (> 250 µm) increased from 70 to 85% in a 10-yr natural fallow on degraded Vertisols.
Southwestern Nigeria[3]	4-yr natural fallow on an eroded and compacted Oxic Paleustalf improved crop yields but the time was not sufficiently long to improve all soil properties.
Nigeria[4]	Growing pheasantwood in former degraded croplands improved microbial biomass, total N, and organic C more than Leucaena.
Austrialia[5]	Replacement of annual crops with native and deep-rooted perennial trees and shrubs restored degraded soils by utilizing the excess water and equilibrating soil-salt-water balance.
Kenya[6]	Leaf biomass additions from six different species of trees and shrubs at 5 Mg ha^{-1} for 5 yr under continuous corn increased organic C and N content.
	No-till
Nigeria[7]	No-till mulched with rice straw at 1, 2, 3, and 4 Mg ha^{-1} yr^{-1} reduced runoff and soil erosion over plowed soils, which had erosion rates six times higher than no-till soils with 4 Mg ha^{-1} yr^{-1} of straw.
Brazil[8]	Combinations of no-till and cover crops in the order: oat/corn, vetch/corn and oat + vetch/corn, + cowpea restored soil fertility on a degraded Paleudult.
	Cover Crops
Nigeria[9]	Legume cover crops increased water retention and infiltration and saturated hydraulic conductivity and reduced compaction on degraded Ultisols.
Bolivia[10]	Legume cover crops contributed to restoration of soil fertility in short natural fallows in semi-arid hillsides between 2,600 m and 3,220 m of elevation.
Brazil[11]	Green manuring with pigeonpea and sunnhep with and without incorporation of the biomass improved the soil fertility on highly degraded croplands.
	Amendments
Southeastern China1[12]	Application of peat at rates of 0, 10, and 50 g kg^{-1} improved the mechanical resilience and increased total porosity of severely degraded Ultisols.
Rwanda, Burundi, and Cameroon1[13]	Animal manure and residue mulch restored soil productivity by reducing soil erosion.
Argentina1[14]	Application of 20 and 40 g kg^{-1} of vermicompost or compost improved soil physical properties and microbial activity, increasing crop yields.
Southern Nigeria1[15]	Application of poultry manure at 10 Mg ha^{-1}reduced soil bulk density and increased water retention capacity and infiltration rates on a degraded Ultisol.
Sub-Saharan Africa1[16]	Application of rice mill waste at rates of 10–15 Mg ha^{-1}increased soil organic matter content of degraded Alfisols

[1]Gafur et al. (2003), [2]Bravo-Garza and Bryan (2005), [3]Kang et al. (1997), [4]Wick et al. (1998), [5]Bell (1999), [6]Nziguheba et al. (2005), [7]Lal (1998), [8]Amado et al. (1998), [9]Obi (1999), [10]Wheeler et al. (1999), [11] De Alcantara et al. (2000), [12]Zhang et al. (2005), [13]Roose and Barthes (2001), [14]Tognetti et al. (2005), [15]Obi and Ebo (1995), and [16]Schulz et al. (2003).

recovery of the soil, enhancing a rapid regeneration of soil properties. Soils under reduced tillage or no-till management have higher resilience because of higher soil organic matter content and activity of soil organisms. Long-term use of conservation tillage restores degraded soils through reduced soil disturbance and high residue input. The benefits of no-till practices for improving soil resilience may, however, depend on the degree of soil degradation, soil type, and climate. No-till management alone may not be enough to recover severely degraded soils. Also, the use of no-till systems in arid and semiarid ecosystems may not be viable if crop residues are either removed or insufficient. Benefits of no-till technology are discussed in Chapter 8.

In general, cropping systems that enhance soil resilience are associated with conservation tillage (Table 16.8). Systems that incorporate legumes and high residue-producing crops are beneficial to improving soil resilience. Restoration of soil is commensurate with the quantity and quality of crop residue input on the soil surface. Cropping systems that leave large amounts ($>5\,\mathrm{Mg\,ha^{-1}}$) of crop residue increase soil organic matter content and percent of water-stable aggregates in the surface horizons. Complex and diverse crops rotations integrated with cover crops are preferable over monocultures to enhance soil resilience. Soil resilience is usually higher under pastures and planted fallow systems than under annual crops. Soil microbiological processes with positive influence on soil resilience are prominent under the improved fallow systems. Pheasantwood is one of the species suitable for soil restoration in tropical ecosystems (Wick et al., 1998). In Australia, woody species such as Eucalyptus are popular to deal with flooding, waterlogging, rising groundwater, and the saline environments (Bell, 1999).

Crop residues, green manures, livestock manures, poultry manure, compost, and other organic amendments are effective strategies to improve soil resilience, especially when managed properly (Table 16.8). Use of organic amendments not only improves the resilience of the soil to mechanical stresses but makes the soil more malleable or friable. Soils amended with organic materials have higher ability to rebound once the mechanical stresses (e.g., tillage operations) are removed. Fresh organic materials (e.g., green manure, manures) revitalize the soil system, activate biological processes of soil fungi and bacteria, and increase the soil's ability to recover. Organic amendments, if available in the amount needed, are more effective than inorganic fertilizers because they improve microbial biomass and activity and nutrient cycling. Organic amendments other than green manures such as animal manure and compost are also essential to restore degraded soils. Manuring in conjunction with minimum or no-till is an effective practice for restoring soils. There are many examples worldwide where application of green manure and organic amendments restored degraded soils (Table 16.8).

Summary

Soils are prone to rapid degradation under intensive cultivation, accelerated erosion, and mismanagement. Most soils are, however, resilient and recover from disturbances under proper restorative measures. This intrinsic ability of soils to counteract degradation is known as soil resilience, which is an essential property to overall agricultural productivity. Soil resilience has paramount implications for crop production, environmental quality, and the projected global climate change. A theoretical definition of soil resilience is complex because of the dynamic and heterogeneous nature of the soil system. Disturbance impacts depend on the soil system and its interactions with management and climate. A degraded soil does not always recover to its original condition but to a level where its performance is not considerably different from that of the pre-degradation level. Severely degraded soils may not recover from disturbance even under refined restorative practices. Deep soils with high organic matter content and high aggregate stability have higher resilience than shallow and unstable soils with low organic matter content.

There are natural (e.g., earthquakes, fires, rainstorms, drought, floods) and anthropogenic (intensive tillage, monocultures, salinization, traffic, grazing) causes of soil disturbance. Soil type, slope, biota, and climate influence the ability of a soil to recover from degradation. The same factors and processes that affect soil formation influence the resilience of soils. Changes in soil properties such as topsoil depth, microbial biomass, organic matter and nutrient content, macroporosity, water-stable aggregates, root biomass, compaction parameters, and salinity and sodicity levels are used as indicators of soil resilience.

Some of the management strategies that promote soil resilience include conservation tillage, residue management, cover crops, manure, integrated nutrient management, conservation buffers, and mechanical practices. Models based on dynamic soil properties, initial condition of soil, equilibrium state, rate of renewal, rate of soil degradation, and management input are used to simulate soil degradation and resilience as well as to estimate the time required for a degraded soil to recover from disturbance.

Study Questions

1. Compare and contrast soil resilience and soil quality, and discuss its importance.
2. Discuss the accuracy of the classification of soil resilience.
3. What are the indicators of soil resilience?
4. Explain the different types of soil disturbance and their impacts on soil resilience.
5. What are the factors of soil physical, chemical, and biological processes affecting soil resilience?
6. Discuss the relationship between soil resilience and factors of soil formation.
7. What are the mechanisms by which conservation tillage promotes soil resilience?
8. Discuss the influence of fallow systems on soil resilience.
9. Discuss the strategies to improve soil resilience.
10. How can soil resilience be measured and modeled?

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Chapter 17
Soil Conservation and Carbon Dynamics

The subject of soil conservation is traditionally discussed in relation to erosion processes and factors, and measures of erosion control. Because of the environmental concerns related to the enhanced enrichment of atmospheric greenhouse gases (e.g., CO_2, N_2O, CH_4), potential of trading of C credits, and continued depletion of soil organic C pool in agricultural and non-agricultural soils with its attendant adverse consequences in soil productivity, the theme of terrestrial C dynamics is also an important global issue. The topic of soil and water conservation is closely related to the dynamics of above- and below-ground biomass C. Improved land use and management systems (e.g., reduced tillage, no-till) used for soil and water conservation are being scrutinized for their potential to influence global C cycle and the ecosystem C budget.

Total soil organic C pool is a sizable component of the terrestrial C pool and its dynamics strongly influence the global C cycle. More C is stored in the soil than either in terrestrial biomass or the atmosphere. Conversion of natural to agricultural ecosystems has caused the depletion of the soil organic C pool across the globe. Excessive tillage, biomass burning and removal, fertilizer application, and extractive farming practices are responsible for about one-fifth of the total annual emission into the atmosphere. The goal is to restore the C pool by increasing the input of biomass C into the ecosystem through reforestation, adoption of conservation tillage practices (e.g., no-till), use of cover crops, manuring, etc. Agricultural practices which enhance C pool are also effective in reducing soil erosion and improving crop productivity. The ecological approach of photosynthesizing CO_2 and converting biomass into soil organic C pool has numerous ancillary environmental and economic benefits in contrast with engineering techniques of injecting of CO_2 into saline aquifers or geologic strata.

17.1 Importance of Soil Organic Carbon

The soil organic C pool is a valuable natural resource. It is the main component of soil organic matter, which moderates physical, chemical, and biological soil processes. Depletion of soil organic C pool exacerbates soil erosion hazard and reduces

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crop productivity. Thus, the soil organic C pool depleted by extractive farming practices and degradation processes must be restored. The soil organic matter stabilizes soil structure, improves soil tilth, promotes root development, increases water retention and nutrient availability, and enhances microbial processes. It reduces soil erosion by stabilizing aggregates and decreasing erodibility, improving water infiltration rate, and reducing the amount and rate of overland flow. It improves water quality by adsorbing and filtering pollutants (e.g., pesticides), which prevent toxic compounds from leaving the sources area and polluting natural waters. Losses of soil organic C pool result in gains in the atmospheric abundance of CO_2 and CH_4. In contrast, soils, through conservation, can be a natural C sink with adoption of recommended management practices.

17.2 Soil Organic Carbon Balance

The soil organic C budget is commonly estimated by computing the difference between vertical inputs and outputs of C while ignoring any lateral component. The vertical inputs include biomass C and organic amendments while the vertical outputs include C emissions and leaching (Fig. 17.1). The lateral components consist of C removed by water, wind, and tillage erosion. The largest fluxes of C occur vertically, but in some soils, lateral flux of C through erosion can also be a significant component. On nearly level landscapes, lateral C flux by water and tillage erosion can be small, but can be large in soils prone to wind erosion. The magnitude of lateral C flux is difficult to quantify because it is influenced by a series of complex and interactive factors such as soil management, landscape characteristics, and climate.

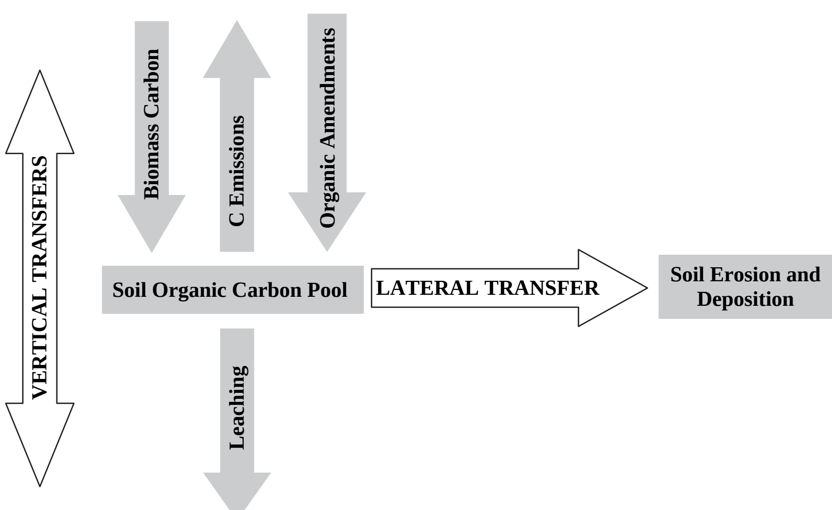


Fig. 17.1 Vectorial transfers of soil organic C pool (After Izaurralde et al., 2007)

A simple model of soil organic C (SOC) balance in the soil along the landscape is shown by Eq. (17.1) (Lal, 2003)

$$\Delta SOC = (SOC_a + A) - (E + L + M) \tag{17.1}$$

where SOC_a is antecedent soil organic C pool, A is C input, E is C transported by erosion, L is C leaching, and M is mineralization of soil organic matter. Leaching of C occurs in the form of dissolved C. Some of the C leached is precipitated in the subsoil and part of it can be transported into large aquatic systems.

17.3 Soil Erosion and Organic Carbon Dynamics

Soil erosion alters the fluxes of soil C because it removes and redistributes the C-enriched sediment and accelerates the process of mineralization (e.g., C emissions). Each process of soil erosion including detachment, transport, distribution, and deposition affects C dynamics. The process of C removal is set-in-motion when the raindrops impact or strike the soil surface. Similar to water erosion, wind erosion removes C in arid and semiarid regions. Indeed, removal of soil organic C by wind can exceed that by water in dry regions with strong wind storms. The amount of C removed by water and wind erosion depends on the magnitude of sediment removal. Surface cover conditions, soil properties, and degree of soil organic matter decomposition are some of the factors that affect the magnitude of C removal. For example, soil and C losses are higher from conventionally tilled soils without crop residue return than from no-till soils with residue mulch. Because of the deposition of C in the bottom perimeter of fields and continued removal of soil from the upper positions of the landscape, the soil organic C concentration decreases in convex positions and increases in the concave or footslope landscape positions. The eroded soil and associated C differ in their characteristics from those of the original or uneroded soil. Sediment transported into depositional areas often contains fine organic and clay particles.

There are six specific processes by which erosion alters C dynamics. These processes are briefly described below:

17.3.1 Aggregate Disintegration

Aggregate breakdown is the very first process by which erosion initiates losses of C pools at the eroding site. Erosive forces such as raindrop impact and shearing force of runoff and wind disrupt, disperse, and slake aggregates. Soil dispersion exposes to microbial attack the soil organic matter-C hitherto protected inside macro- and micro-aggregates. Thus, the process of C release by erosion is initiated when soil aggregates are disintegrated by the erosive forces. The systematic reduction in size of secondary soil particles with erosion concomitantly results in a systematic release of more occluded C. While the process opposite to disintegration (e.g., aggregation) promotes encapsulation of soil C, aggregate disruption leads to release of C and to its microbial decomposition.

17.3.2 Preferential Removal of Carbon

Disintegrated aggregates and primary particles are small in size and readily transported by water and wind. Erosion is a selective process. Because of the low density of the soil organic matter, C-enriched soil particles are more easily removed by water and wind than the more dense and compact inorganic particles. Also, lighter soil particles are also transported to longer distances than heavier particles. Another factor by which soil organic C is preferentially removed by water and wind is its location within the soil profile. The organic C is mostly concentrated in the upper few centimeters of the soil surface, which is the region of active perturbation by erosional processes.

17.3.3 Redistribution of Carbon Transported by Erosion

The C removed by erosional processes is redistributed all over the landscape. Some of it is deposited in field depressions or transported off-site to rivers, lakes, and eventually to oceans. Depositional areas normally have greater concentration of soil organic C than the eroding or convex sites of the landscape. Land topography, degree of erosion, and site-specific soil characteristics define magnitude of C removal by erosional process and the distance to which it is transported.

17.3.4 Mineralization of Soil Organic Matter

The soil organic matter transported by erosion comprises the labile C pool, which is rapidly oxidized (Fig. 17.2). While the labile C in uneroded soils is also susceptible to rapid decomposition, erosion accelerates this process by shifting and mixing the

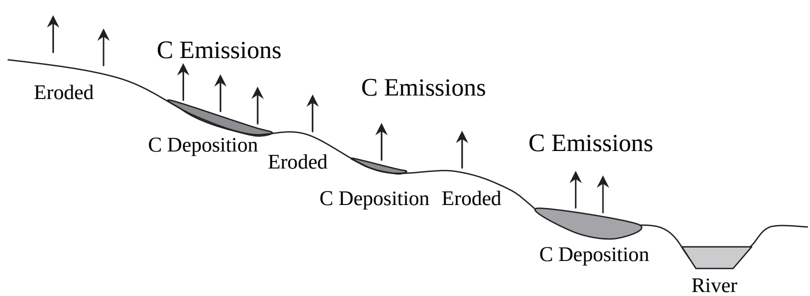


Fig. 17.2 Dynamics of soil organic C during erosion (After Lal, 2003)

soil, disintegrating the aggregates, and altering temperature and moisture regimes. Significant mineralization of C occurs at all stages of soil erosion. The C emissions are highly variable among the different erosion phases, impedes an accurate quantification of the net emissions from eroding landscapes.

17.3.5 Deposition and Burial of Carbon by Transported by Erosion

The soil organic matter at the depositional sites is subject to rapid anaerobic decomposition (e.g., methanogenesis, mineralization), causing loses of CO_2, CH_4, and N_2O (denitrification) gases. The soil organic matter deposited in the upper 0- to 20-cm soil depth is particularly rapidly mineralized. Emissions of C from the depositional areas can be higher than at the eroding sites because the eroded sediment is enriched with soil organic C, and detachment and dispersion of aggregates expose physically and chemically protected C to microbial processes. Some of the eroded soil organic matter buried in deeper layers does not decompose easily and thus promotes long-term C sequestration. A portion of the buried C forms stable compounds (e.g., calciferous compounds) and re-aggregation with low rates of mineralization.

17.4 Fate of the Carbon Transported by Erosion

The fate of eroded C is rather complex and uncertain. A simple model to estimate the fate of eroded C is as follows (Lal, 2003):

$$SOC_f = (SOC_a) - (C_i + C_t + C_d + C_r) + (C_b + C_w) \tag{17.2}$$

where SOC_f is soil organic C, SOC_a is initial amount of soil organic C, C_i is amount of C oxidized *in situ*, C_t is C oxidized during transport, C_d is C oxidized in depositional zones, C_r is C oxidized in aquatic systems, C_b is C buried in depositional zones, and C_w is C buried in aquatic systems. The major uncertainly lies with the amount of C emitted during erosion. Some estimates show that most of the C transported by erosion is redistributed along the landscape (Fig. 17.3), but about 20% of C transported by erosion is emitted. Depending on the magnitude of erosion, the largest of amount of C (0.4 Pg C yr^{-1}) would be emitted in Asia in direct proportion to its high rates of soil erosion (Fig. 17.4).

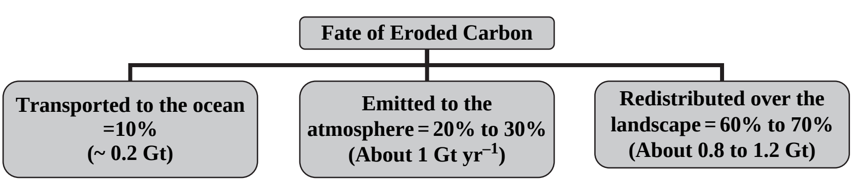


Fig. 17.3 Distribution of C transported by erosion (After Lal, 2003)

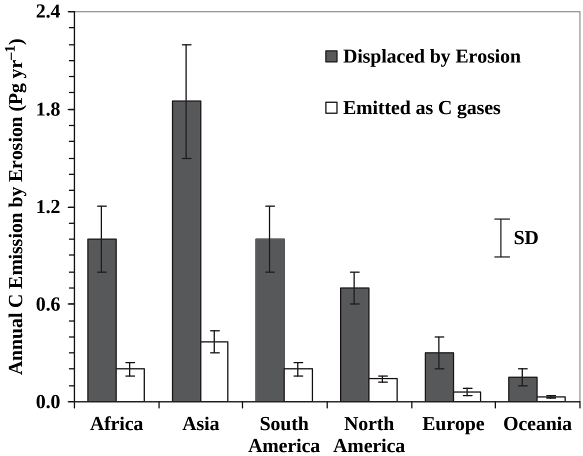


Fig. 17.4 Soil organic C transported by water erosion (After Lal, 2003)

17.5 Carbon Transported by Erosion: Source or Sink for Atmospheric CO_2

There are two main contrasting views about whether the C transported by erosion is source or sink of atmospheric CO_2:

1. The first view is that burial of C transported by erosion with sediments at lower landscape positions protects the C from losses, and thus erosion constitutes a sink of atmospheric CO_2 and a potential means for long-term C sequestration. This view also assumes that C lost from the croplands is easily replaced by biomass production, and that mineralization of soil organic matter-C transported by erosion is negligible. Soil erosion is a sink of about 26% of C transported by erosion in agricultural soils (Van Oost et al., 2007).
2. The second view considers, in contrast, that C transported by erosion is a source of atmospheric C loss and is supported by the following arguments: One, most of the soil organic matter transported by erosion is rapidly oxidized during transport and deposition because it consists primarily of labile organic matter fractions. Two, the soil organic C lost by erosion is not easily replaced. Erosion reduces biomass production at the eroding site and thus input of new C, which is crucial to replacing the C transported by erosion, is reduced. Eroded soils have shallow topsoil layers with limited amount of plant available water and nutrients required for plant growth. Continued erosion causes a downward spiral of reduction in soil organic C pool. Biomass C input may replenish the C transported by erosion in slightly and moderately eroded soils, but it is likely to be insufficient to match large losses of C in severely eroded soils.

These controversies warrant a clarification of whether agricultural erosion is a sink or source of atmospheric CO_2. Reliable data on C export and deposition by all forms of erosion including water, wind, and tillage are needed to elucidate the mixed perceptions. It is, however, clear that C transported by erosion from sloping landscape positions is not readily replaced as the highly eroded sites produce lower biomass yields than uneroded sites. The continued erosion would result in increasingly lower biomass production and C input into the system. Quantification of C mineralization during transport and at the depositional sites is important to understand the erosion-induced dynamics in C cycle.

17.6 Tillage Erosion and Soil Carbon

Dynamics of C need to be discussed not only in regards to water and wind erosion but also in relation to tillage erosion. Tillage erosion can cause the redistribution of C along cultivated landscapes similar in magnitude to that caused by water and wind erosion. The main difference is that C transported by tillage erosion is mostly accumulated in the lower portion of the fields until water and wind erosion intervene. Thus, combined forces of tillage, water, and wind erosion cause off-site transport of C. Frequent plowing loosens up and moves the soil downslope and predisposes the soil to removal by water and wind. Tillage erosion also exposes subsoil horizons with low concentration of C, and the original topsoil is moved downslope where it accumulates to form stratified C-enriched soil deposits. Tillage erosion reduces C concentration in the upper field positions and increases it in the lower positions when compared to control (no-till) soils (Fig. 17.5).

Exposed subsoil horizons in shoulder slopes normally have higher clay content and lower soil organic C concentration due to tillage. Tillage erosion translocates all forms of C unlike water and wind erosion which is a selective process and mostly removes fine lighter particles. Higher soil organic C concentration in the subsoil horizons within the buried soil profiles is not uncommon. The buried C can have important implications to long-term C sequestration. Residence time of C increases with the burial of C. This process increases long-term C sequestration in sloping cultivated lands. A portion of C transported by tillage can be also a net C source because it is easily mineralized and emitted into the atmosphere. Use of tracers such as ^{137}Cs and ^{210}Pb (ex) resulting from the fallout of radionuclides are methods used for tracking C redistributed by tillage over the landscape.

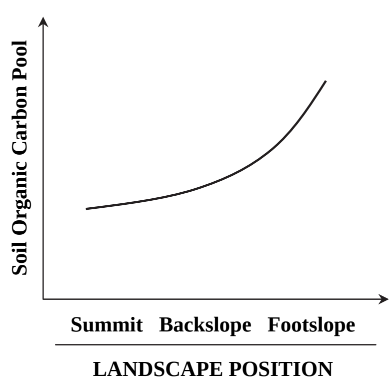


Fig. 17.5 Typical redistribution of C by tillage erosion in sloping soils

17.7 Conservation Practices and Soil Organic Carbon Dynamics

Increasing soil organic C concentration through conservation practices is a key to controlling soil erosion and improving productivity. Improved conservation practices can store C more than traditional tillage practices while reducing soil erosion. Thus, they can accomplish two goals: reduce runoff and soil loss, and increase C storage and reduce C emissions. Some practices which enhance soil organic C pool are summarized below:

17.8 No-Till and Soil Carbon Sequestration

Excessive tillage practices disturb soil and cause rapid oxidation of soil organic matter, thereby increasing flux of C to the atmosphere. It is estimated that as much as 60% of soil organic C in cropland soils of the temperate regions and 75% in the tropics has been depleted by excessive tillage and erosion (Lal, 2004). As a result, no-till farming is being considered as a promising alternative to plow tillage for reducing and restoring C as an ancillary benefit. Because of reduced soil disturbance and return of crop residues as mulch, no-till systems can promote long-term C sequestration. Greater C sequestration in no-till than in plow tillage can off-set emissions by fossil fuel combustion and reduce risks of global climate change. The C accumulation in some no-till soils within the surface horizons can be two to three times higher than that in tilled soils (Table 17.1).

Table 17.1 Influence of tillage systems on soil organic carbon pool on mass per area basis ($Mg\ ha^{-1}$) for selected soils in the surface layers (<30 cm depth).

Soil	Management duration (yr)	Moldboard plow	Chisel plow	No-till
Silt loam[1]	Corn, >15	19.6	19.4	36.3
Clay[2]	Corn, 12	21.5		22.9
Silt loam[3]	Corn, >15	24.0	21.6	38.7
Slit loam[4]	Wheat Residue, 10	10.3		11.5
Silt loam[5]	Corn, 28	17.7	24.1	45.4

[1]Blanco-Canqui et al. (2005), [2] De Assis and Lancas (2005), [3]Blanco-Canqui et al. (2004), [4]Duiker and Lal (1999), and [5]Lal et al. (1994).

17.8.1 Mechanisms of Soil Organic Carbon Sequestration

The C sequestration in the soil is a function of several factors including soil structural stability. Soil aggregation is the nucleus of numerous mechanisms of long-term C sequestration. Soil aggregates occlude C physically and reduce mineralization of soil organic matter. No-till management stabilizes and protects soil aggregates, which are essential to long-term C sequestration. Residue mulch is a source of energy, shelters soil organisms and increases their activity, resulting in greater macro- and micro-aggregation. Thus, no-till soils are rich in aggregate-forming microbial biomass including bacteria, fungi, and mycorrhizal fungi. The C-rich residues in aggregates are protected from microbial and enzymatic actions, thereby remaining relatively undecomposed for a long time.

17.8.2 Excessive Plowing

Intensive plowing is a primary factor that exacerbates depletion of the soil organic C pool because it breaks aggregates, exposes C to microbial processes, and accelerates turnover of C-enriched aggregates. It impedes natural formation of stable micro-aggregates ($<250\,\mu m$) and macro-aggregates ($>250\,\mu m$) because it accelerates the decomposition of organic binding agents responsible for stabilizing and arranging the aggregates. Any cultivation practice that reduces the disruption of aggregates protects soil organic C pool. Intensive plowing thus leads to the formation of C-depleted microaggregates. Tilled soils not only create a less conducive environment for natural aggregation but cause frequent instability of the existing aggregates. Because of low aggregate turnover, no-till soil maintains more soil organic C in surface than tilled soils. Similarly, no-till soil may have as much as twice more stable and free microaggregates than plowed soils, promoting C encapsulation within the microaggregates due to slower macroaggregate turnover.

17.8.3 Site Specificity of Carbon Sequestration

While no-till farming conserves soil and water and reduces production costs, its potential for sequestering SOC is, however, site specific (Blanco-Canqui et al., 2008). The C accumulation depends on many factors including soil texture, internal drainage, specific soil management, agro-ecosystem, and climatic conditions. The perceived benefits of no-till to C sequestration must not be generalized to all soils. Sloping and erosion prone soils benefit the most from the adoption of no-till technology. The no-till systems may not always increase C storage as compared to plow tillage in clayey and poorly drained soils and cold climates. Occasional plowing or subsoiling may be needed in clayey and compact soils to enhance root proliferation. Adoption of no-till alone without the companion practices such as cover crops, high biomass producing crops, crop residue mulch, complex crop rotations (legumes or perennial grasses), and manuring may not be any better than plow tillage for C sequestration and soil productivity improvement.

17.8.4 Stratification of Soil Carbon

There is a strong stratification of C in the upper layers in no-till soils due to surface residue mulching. Thus, beneficial impacts of no-till on soil organic C accumulations are often limited to the upper few centimeters (<10 cm) of the soil surface

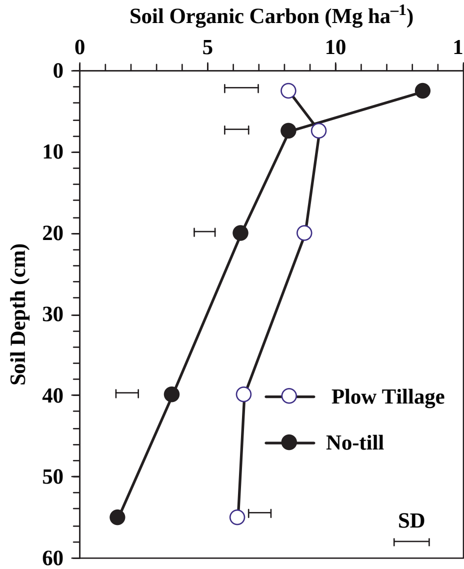


Fig. 17.6 Soil organic C pool as influenced by long-term tillage management on a silt loam (After Blanco-Canqui et al., 2008). While no-till management generally favors greater soil organic C accumulation in the surface layers (<10 cm) compared to plow tillage, the opposite can be true in deeper layers

where most residues are concentrated (Fig. 17.6). The compacted soil surface in some no-till systems tends to confine C additions to the upper few centimeters and create stratification of C because of limited incorporation of fresh organic residues to deeper layers (Franzluebbers, 2002).

17.8.5 Soil-Profile Carbon Sequestration

Some researchers have argued that the higher C sequestration in no-till systems reported in numerous studies may simply be due to the shallow sampling protocol (Baker et al., 2007). The fact is that most of the studies reporting higher C sequestration in no-till than in plowed soils have based their conclusions from soil samples obtained for the shallow depth (0- to 30-cm) (Table 17.1). The few studies reporting C sequestration for the whole soil profile have observed either no statistical differences in C below 30 cm depth or even lower C in no-till relative to plowed soils. A regional study of soil-profile C distribution (0 to 60 cm) under no-till conducted across 11 soils in the eastern U.S. showed that C concentrations in no-till soils were greater than those in plowed soils only in five out of 11 soils in the 0- to 10-cm depth (Blanco-Canqui et al., 2008). The same study showed that the total C for the whole soil profile between no-till and plow tillage did not differ. In fact, plow tillage stored more C than no-till in three soils for the whole soil profile. Results of this regional study are not, however, conclusive because the no-till and plowed fields were not always managed under the same cropping systems. In some soils, the lower C in plowed soils as compared no-till farming in the upper layers can be compensated by C gains in deeper soil. Similar results were reported in other soils (Yang and Wander, 1999; Puget and Lal, 2005).

There are three reasons by which plow tillage may increase soil organic C in deeper soil:

1. ***Annual burying of crop residues with plowing in contrast with no-till where residues are left on the soil surface.*** Crop residues buried by plowing increase input of C into the subsoil.
2. ***Differences in the rooting depth of crops between no-till and plow tillage.*** Low bulk density in plowed soils can favor penetration of roots to 50 cm depth. This rooting depth is higher than the typical soil sampling depth for C analysis. The relatively greater compaction in no-till soils may limit root growth to the upper layers.
3. ***The buried crop residues in plowed soils are in close contact with soils as compared to residue mulch in no-till soils.*** The closer association can favor greater C protection and lower decomposition rates in deeper layers of plowed soils. It can also promote formation of recalcitrant compounds or stable C with longer residence times (Six et al., 1998). Mechanisms of C protection and stabilization between no-till and plowed soils must be further studied to elucidate the potential of no-till for long-term C sequestration.

Data available to this point suggest that increases in C concentration by no-till farming depend on soil type, management, and climate. Soils in which no-till stores more C than plow tillage, the gains in C are solely confined to the upper layers ($<$20 cm). Data also suggest that the potential of no-till farming for sequestering C is more complex than hitherto perceived. No-till technology is an unparalleled system for reducing soil erosion and reducing tillage costs, but the view that it can sequester C more than plow tillage in all soils needs an objective assessment based on measured data. A detailed quantitative analysis of the potential of no-till for C sequestration must be conducted at local and regional levels encompassing a wide range of soils and ecosystems.

17.9 Crop Rotations

Complex and diverse crop rotations increase soil organic C pool over monocropping especially if rotations leave large amounts of crop residues. The soil C accumulations in two historical agricultural experimental fields of the world decreased with continuous monocropping of corn by about 56% in Sanborn Field and by about 45.6% in Morrow plots in Illinois (Aref and Wander, 1998; Rachman et al., 2003). Changes in C concentration in these long-term experiments were small in recent decades, indicating that the soil system is near equilibrium conditions. Using grass species and diversifying crop rotations can lead to the highest increases in soil organic C concentration and pool.

The soil organic C concentration increases with increase in cropping frequency especially under no-till (Karlen et al., 2006). For example, elimination or reduction of fallow systems is an important option for enhancing soil organic C and improving soil fertility. Practices which maximize crop production and provide abundant biomass must be implemented to minimize losses of soil organic C pool. The soil organic C concentration is a function of crop residue input. Crop rotations combined with no-till are particularly important to increasing soil organic C pool. For example, soil organic C pool under potato, which is a highly intensive cropping system and produces little residue, can be increased if alternated with high biomass-producing crops. Gains in soil organic C concentration are higher with increasing diversity and duration of crop rotation cycles.

17.10 Cover Crops

One of the direct benefits of growing cover crops is input of biomass C into the soil. Cover crops play a major role in revitalizing the soil and reducing erosion while ensuring buildup of soil organic C. The fresh and abundant C input by cover crops is vital to soil because of the importance of soil organic C to soil physical, chemical, and biological properties. Unlike main crops, the totality of plant materials is returned to soil, increasing soil organic C pool, recycling nutrient, and enhancing soil functions. Increases in soil organic C concentration by cover crops can be small but consistent (Table 17.2). The magnitude of increase in soil organic C pool depends on the cover crop species, soil type, tillage method, biomass management (e.g. hay, mulch, green manure), and climate. Increases in soil organic C concentration by cover crops in no-till soils may not always be significant depending on site-specific characteristics. Harvesting cover crops as hay reduces their benefits for increasing soil organic C concentration.

Table 17.2 Changes in soil organic C concentration with the establishment of cover crops

Soil	cover crop	Soil organic C ($g\,kg^{-1}$)	
		No cover crop	With cover crop
Silty clay loam[1]	Annual ryegrass, fall rye, and spring barley	17	19
Clay loam[2]	Italian ryegrass and white clover	30	32
Sandy clay loam[3]	Wild kulthi, centro, calapo, and kudzu	4	8
Silt loam[4]	Rye	15	16

Source: [1]Liu et al. (2005), [2]Yang et al. (2004), [3]Dinesh et al. (2004), and [4]Kuo et al. (1997).

17.11 Crop Residues

Leaving crop residues on the soil surface increases soil organic C pool while reducing soil erosion (Table 17.3). Plowing under residues may accelerate its decomposition through the action of soil organisms. It is, however, the soil texture, drainage, tillage, and climate that control the residue decomposition rates. Response of soil organic C to residue management can be slow in clayey soils. No-till under

Table 17.3 Changes in soil organic C concentration with residue removal

Soil	Soil slope (%)	Depth (cm)	Tillage system	Crop	Residue rate (Mg ha^{-1} yr^{-1})	Organic carbon (Mg ha^{-1})	
						No residues	With residues
Silt loam[1]	6	5	No-till	Corn	5	14	19
Silt loam[1]	2	5	No-till		5	15	20
Clay loam[1]	<1	5	No-till		5	16	18
Silt loam[2]	1	10	No-till	Wheat	8	9	13
Silt loam[2]	1	10	Ridge Tillage		8	9	14
Silt loam[2]	1	10	Plow Tillage		8	8	11

Source: [1]Blanco-Canqui and Lal (2007) and [2]Duiker and Lal (1999).

continuous corn accumulates soil organic C more than no-till under corn/soybean rotations because of greater biomass C input from corn. Quantity of residue inputs in interaction with soil texture defines the pool, turnover, and retention of C. Soybean residue has a lower C:N ratio than corn residues and is thus easily decomposed. Leaves and husks of corn residue decompose faster than cobs and stems. Residue accumulated in surface layers (e.g., no-till) decomposes slowly and promotes long-term storage of C. Residue removal for expanded uses such as biofuel production has a direct adverse impact on soil organic C pool because C pool is a function of the amount of crop residue left on the soil. Removal of crop residues affects soil organic C dynamics by: (1) directly reducing the C content, (2) increasing soil erosion and thus C losses, and (3) reducing biomass needed to maintain C pools.

17.12 Manure

Manure application increases soil organic C concentration and reduces runoff and soil erosion. The soil organic C concentration increases linearly with increase in rate of addition of manure although changes in soil organic C concentration at high rates of addition may be small. Manure application enhances formation and stabilization of aggregates and promotes long-term C storage. It increases both the particulate organic matter and mineral associated soil organic C in macroaggregates. It also reduces runoff and soil losses by stabilizing aggregates and increasing water infiltration. No-till management slows the mineralization of manure unlike tilled systems. The soil organic matter in manure entering the soil is initially in the labile form and then converted into mineral associated C pool. Well-planned management of manure is vital to soil organic C sequestration in agricultural lands. Manured no-till soils have significantly higher soil organic C pools than no-till soils without manure (Fig. 17.7). Since gains in soil organic C in no-till soils without manure are often confined to the upper 10 cm of soil surface due to residue stratification, no-till in combination with manuring is a useful strategy to increase soil organic C pool in deep soil layers (Fig. 17.7).

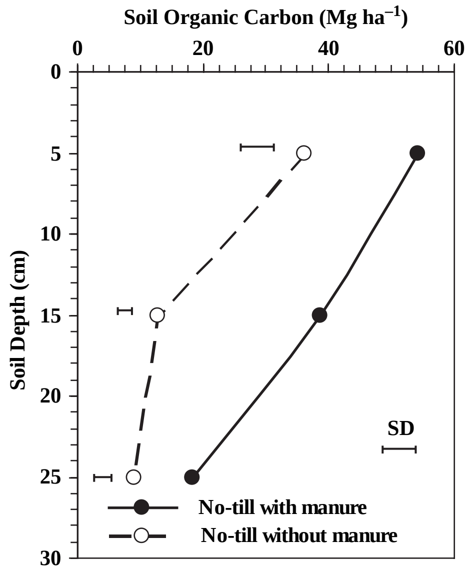


Fig. 17.7 Depth distribution of soil organic C for no-till with and without manure in a sloping silt loam (After Blanco-Canqui et al., 2005). Manuring combined with no-till can increase soil organic C pool by several orders of magnitude as compared to non-manured soils. The effects of no-till and manuring on increasing soil organic C pool are additive

17.13 Agroforestry

Agroforestry systems play a crucial role in increasing soil organic C storage and reducing net emissions of greenhouse gases in addition to soil erosion control. Because of their rapid growth and abundant above- and below-ground biomass, trees in an agroforestry system have the ability to increase and stabilize soil organic C pool. Agroforestry practices can recapture some of the soil organic C that was lost with land clearing and biomass burning. Enhanced soil organic C sequestration by agroforestry systems can be a tradeoff for the land taken out of production for establishing agroforestry trees and/ or shrubs. The amount of soil organic C regained by the reintroduction of agroforestry practices may not equal to that lost in the clearing of primary forests. Thus, long-term agroforestry systems with mixed plant species are more effective for soil organic C storage than systems with single species. Mixed plant species allows some harvesting of trees and still maintain their potential to store soil organic C. Managing the stands is critical to soil organic C storage in agroforestry systems. Complete removal of trees particularly in short rotation stands offers little benefit to long-term C storage and this type of management may not store C more than that under seasonal crops.

At present, the amount of soil organic C stored by agroforestry systems can not be accurately quantified because of the lack of: (1) information on the land area under agroforestry systems around the world and (2) a complete understanding of above- and below-ground biomass C storage in trees and/or shrubs. These two problems are confounded with related factors affecting soil organic C storage including differences in tree species, age of species, management, agroforestry practice, ecosystem, and climate. Fertilization, irrigation, and weed and pest control in alley cropping generally favor soil organic C storage but pruning or thinning may reduce increases in soil organic C storage particularly in sparse stands. Vegetation with dense root system in alley hedgerows may, however, have a high capability for below-ground C enrichment.

Agroforestry systems can store between 12 and 228 Mg ha^{-1} of C with an average of 70 Mg ha^{-1} and thus, based on the total area in the world that is suitable for agroforestry (585–1215 $\times$ 10^{6} ha), between 1.1 and 2.2 Pg of C can be stored (Dixon, 1995). Agroforestry under temperate (63 Mg ha^{-1}) and humid regions (50 Mg ha^{-1}) stores C more than in sub-humid (21 Mg ha^{-1}) and semi-arid (9 Mg ha^{-1}) regions (Schroeder, 1994). The transformation of degraded or substandard soils to agroforestry is an important strategy to sink C. The estimated annual C accumulation under various agroforestry systems in the USA by 2025 is about 74 Tg C for alley cropping, 9 for silvopasture, 4 for windbreaks, 2 for forest farming, and 1.5 for riparian buffers (Montagnini and Nair, 2004).

Implications of traditional forest systems on C storage and soil improvement are well known, but those of agroforestry systems are not. The soil organic C storage rates vary across regions, depending on the type of agroforestry practice, management, and climate. Most of the estimates of soil organic C storage by agroforestry systems are for tropical regions and rates range between 0.1 and 3.9 Mg ha^{-1} yr^{-1}. Silvopastoral systems and alley cropping systems store the greatest amount of C.

17.14 Organic Farming

The soil organic C concentrations under organic farming may increase, decrease, or remain unaffected relative to the conventional farming systems (Drinkwater et al., 1998; Green et al., 2005). Reasons for these inconsistencies include a wide range of factors such as tillage practices, source of C, cropping systems, climate, and soil properties.

17.14.1 Excessive Tillage

Excessive tillage such as moldboard plowing reduces the potential benefits of organic farming to increase soil organic C concentration that might be expected from the addition of organic amendments. It destroys and mixes residues and accelerates their decomposition, reducing soil organic C storage in contrast to no-till farming in which the residues left on the soil surface are subject to slow decomposition. Losses in C from organic farming with excessive tillage can be therefore as high as those from conventional farming, negating any benefits of organic farming. Combination of organic farming with no-till and reduced tillage is a useful strategy to enhance soil C sink capacity.

17.14.2 Source of Soil Organic Carbon

Organic farming with short crop rotations and limited manure application may have lower soil organic C pools than conventionally farmed soils with extended rotations and annual additions of animal manure. Manure application, a common practice in organic farming, increases C storage, nutrient pools, and biological activity. Addition of animal manure combined with legume-based crop rotations increases soil organic C storage especially in conjunction with reduced tillage and complex crop rotations. Green manures from cover crops also increase soil organic C storage. Recycled urban waste products or composted and uncomposted local organic amendments are rich in organic matter and nutrients. In some soils, losses of soil organic C as CO_2 emissions can also be higher in organic farming than in conventional farming from the heavy addition of animal manure and excessive tillage for weed control.

17.14.3 Cropping Systems

Organic farming systems that incorporate complex crop rotations and cover crops increase soil organic C concentration when tillage operations are reduced. Crop rotations in organic farming break pest cycles, suppress proliferation of certain weeds, and reduce the need for excessive tillage. Rotations have long been used to control weeds even prior to the introduction of pesticides and herbicides. Rotations are essential to suppressing specific weed species by increasing competition. Rotations must include a variety of crops to effectively break the weed life cycles. Planting legumes (e.g., alfalfa), grasses and perennial crops or winter covers disrupts the weed cycles while increasing soil organic C storage.

17.15 Bioenergy Crops

Growing bioenergy crops, such as perennial warm season grasses and short-rotation woody perennials, conserves water and soil while increasing soil organic C concentration and reducing C emissions. In conjunction with some of the above-ground biomass returned, the below-ground biomass enhances soil organic C pools more than growing short-statured grasses and row crops. The deep root system of bioenergy crops enhances accumulation of soil organic C in deeper layers. When warm season grasses (e.g., switchgrass) are cut, the height of cut should be ≥ 15 cm to maintain proper surface cover. The soil organic C storage by bioenergy crops is generally high and rapid in soils with low antecedent soil organic C pool. Growing warm season grasses can be particularly feasible in marginal soils to reducing the competition for land with row crops.

Switchgrass, one of the common warm season grasses, has been traditionally used as grass barriers or buffers for soil and water conservation. This warm season grass species has also the potential to sequester C between 0.8 and 1.0 Mg C ha^{-1} yr^{-1} in plant and litter biomass (Tufekcioglu et al., 2003). The capacity of switchgrass to sequester C can be higher than cool season grass and row crops. Switchgrass and other warm season grasses enhance belowground biomass storage because of their high root biomass, which is essential to long-term C sequestration. Magnitude of increases in soil organic C pool depends on soil type, soil management, and nutrient status.

17.16 Reclaimed Lands

Agricultural soils are degraded because of the loss of soil organic C pool, which reduces soil resilience, biomass production, and filtration of pollutants, and increases risks of soil erosion. Reclamation of degraded soil is a viable pathway to sequester soil organic C in addition to that sequestered in productive lands. Degraded soils are potential C sinks because their C contents are below the saturation levels. Adoption of conservation tillage restores degraded soils and enhances C sequestration. The most common practices to restore/enhance soil organic C sequestration are: (1) addition of manure and crop residues, and (2) establishment of pasture. The efficiency of the practices varies with ecosystem and climate (Table 17.4). In some soils, organic inputs often mineralize in a short time and negate any consistent increases in soil organic C pools.

Drastic land disturbance during mining operations causes losses of soil organic C pool due to rapid soil organic matter decomposition. Soil disturbance reduces aggregation and microbial biomass and activity responsible soil organic C sequestration and protection in the soil. Revegetation is the path for C accretion in reclaimed minesoils. What has been lost as CO_2 has to be replaced with C input through above- and below-ground biomass input provided that C input is higher than C losses. The C sequestration in minesoils increases with time (Fig. 17.8A) following reclamation until a new equilibrium is reached (Fig. 17.8B). Soil structure development during reclamation mediates the sequestration of soil organic in the soil. Reclamation of minesoils with growing vegetation is a potential alternative for increasing soil organic C pools in terrestrial systems. Proper reclamation rapidly increases C storage and improves soil structure. In some reclaimed soils, rates of soil organic C storage may not only catch up those of unmined sites but may surpass the original levels of soil organic C pool, depending on management and vegetation type (e.g., high-biomass producing plants).

Table 17.4 Changes in soil organic C with restorative practices

Soil	Management and duration	Soil organic C (g kg^{-1})	
		Before restoration	After restoration
Silty-clay[1]	Leguminous woody species, 5 yr	13	15
Sand loam to sandy clay loam[2]	Natural fallow, 8 yr	13	14
Silt loam[3]	Reforested, 5 yr	7	13
Sandy clay loam[4]	Grasses/legumes, 5 yr	7	9

Source: [1]Bravo-Garza and Bryan (2005), [2]Atsivor et al. (2001), [3]Islam and Weil (2000), and [4]Obi (1999).

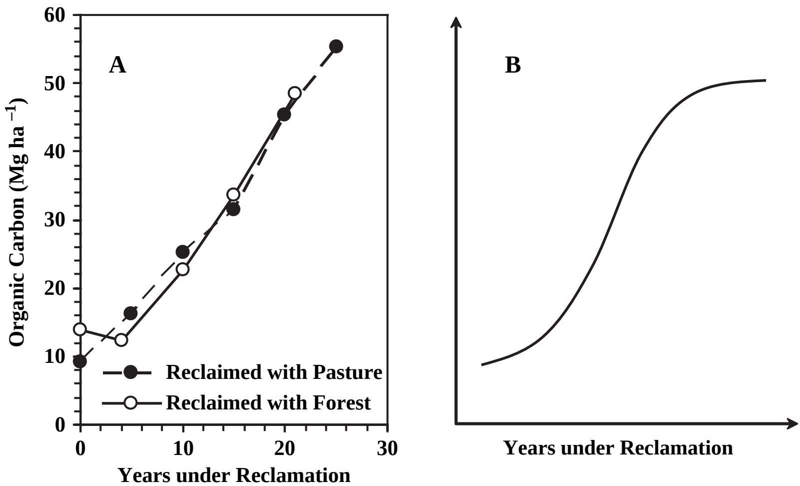


Fig. 17.8 Storage in soil organic C in reclaimed minesoils with time (After Akala and Lal, 2001)

17.17 Measurement of Soil Carbon Pool

The soil organic C concentration must be accurately measured to build reliable C inventories and assess the dynamics of terrestrial C. The Walkley-Black and dry combustion are common laboratory techniques used to measure the C concentration in soil samples. The analysis of stable C isotope ($\delta^{13}C$) by mass spectrometry is another tool to assess C dynamics. The $\delta^{13}C$ is an in situ marker that allows the monitoring the C turnover rates over time. For example, because corn is a $\delta^{13}C$ enriched plant, the analysis of the $\delta^{13}C$ permits the tracing and separation of corn-derived C from the relic or old C of a previous crop with a different isotopic signature. The conventional methods of C analysis (e.g., dry combustion) involve soil disturbance and require extensive soil sampling and sample preparation for analysis. Several emerging methods provide additional tools including Laser induced Breakdown Spectroscopy (LIBS), Inelastic Neutron Scattering (INS), infrared reflectance spectroscopy (IRS), and remote sensing (Gehl and Rice, 2007). These methods are expected to reduce the intensive sampling and preparation, provide a rapid analysis of C, and allow the *in situ* determination of C.

17.17.1 Laser Induced Breakdown Spectroscopy (LIBS)

This method is based on the atomic emission spectroscopy and consists of focusing a laser pulse on a small intact soil core, collecting light emitted by the sample in a spectrograph and detector, and relating the light intensity to the total C measured using the conventional techniques. While this method is still under development, it shows promise to detect C concentration within 300 mg of error with an accuracy of 3 to 14% (Cremers et al., 2001). It is rapid because it provides a reading in < 60 s. Interference of plant roots, partly decomposed plant residues, inorganic C, soil texture (e.g., silicon content), and soil water with the LIBS measurement must be resolved prior to a large-scale use of LIBS.

17.17.2 Inelastic Neutron Scattering (INS)

This is a new technique to non-destructively and non-invasively measure soil organic C. It is based on inelastic scattering of fast 14 MeV neutrons from C nuclei in the soil and subsequent measurement of the emitted 4.44 MeV gamma rays (Wielopolski et al., 2000). The peak intensity of C is obtained from the gamma rays and is related to the soil organic C concentration in the soil. The *in situ* INS equipment consists of a neutron generator and a detector properly shielded to eliminate radiation hazards. It can detect C concentration within 5 to 12% of error. This technique is also under development.

17.17.3 Infrared Reflectance Spectroscopy (IRS)

Infrared reflectance spectroscopy (IRS) is another emerging technique that can be used for the estimation of soil organic C concentration under lab or *in situ* conditions (Brunet et al., 2007). Mid-infrared (MIR) and near-infrared (NIR) regions have been used to predict C concentration based on the reflectance signal of soil C. Each atom in the soil has a specific reflective property in the visible/near infrared zone (between 800 and 2500 nm of wavelength). Since C is mixed with the soil, its specific signature can not be completely isolated. Thus, calibration of the reflectance signal with measured C is essential to the use of IRS. A drawback of the IRS approach is the need of preparing soil samples and using homogeneous sets of soils for improving accuracy. Ground soil samples (0.2 mm) provide more accurate estimates of C than unground samples (sieved through 2 mm). Accuracy is also improved when soil samples are separated and analyzed by textural differences. Statistical parameters such as the partial least-squares (PLS) analysis, principal component analysis (PCA), and pedotransfer functions (PTFs) are being used to improve the calibration between the infrared spectra and measured C with variable outcomes.

17.17.4 Remote Sensing

Satellite radar imagery and ground truth are increasingly being used for monitoring changes in land use and management. The remote sensing tools do not directly measure soil C but permit the collection of site-specific information on surface cover (e.g., growing vegetation, residue cover, bare soil), which can be correlated to biomass C input. Because soil organic C is mostly concentrated near the soil surface, aerial photographs, and electromagnetic radiation are used to estimate C based on soil color and the dark color of soil organic matter. This approach provides, however, qualitative information and may be effective only for soils with high soil organic C concentrations. Relationships between soil organic C concentration and soil reflectance are often masked by soil surface color (e.g., parent material, soil water content, soil texture, and chemical properties, surface residue cover). Now, remote sensing is being combined with GIS to produce maps of land cover and collect input data for C predictive models. Higher reflectance resolution is required to estimate small variations in C concentration.

17.18 Soil Management and Carbon Emissions

Soil management influences emissions of greenhouse gases, depending on soil type and tillage management. Soil conservation practices such as no-till and residue management systems sequester C and can thus reduce C emissions. On the contrary, excessive tillage increases C emissions over no-till systems especially immediately after tillage because it aerates the soil, activates microbial processes, and accelerates the decomposition of organic materials (Reicosky and Archer, 2007). The deeper the plowing, the greater the C fluxes from soil. Residue management also influences C emissions. The crop residue mulch left on the soil surface may reduce C emissions by constituting a physical barrier to C fluxes, by reducing soil temperature, and by not being mixed with the soil matrix. Tillage practices that leave no residues on the soil surface generally have greater C fluxes. Blanco-Canqui et al. (2007) reported, however, no significant differences in CO_2 fluxes among plots with different levels of corn residue cover across various regardless of significant differences in soil organic C pool among plots.

Weather conditions and site-specific characteristics can affect the rate of C fluxes. In dry years, emissions of C from heavily mulched no-till soils may be greater than those from those without mulch due to higher water content, higher microbial activity (e.g., earthworms), and more favorable soil temperature in mulched soils. Thus, C emissions depend on the soil type, quantity of crop residues, and weather conditions. The C emissions are also subject to temporal and spatial variability. Emissions are normally higher in spring and summer than in late fall and winter months due to the higher soil temperature, microbial activity, and plant growth.

The C emissions are measured using open and closed chamber methods based either on mass balance or gas diffusion theory. Automated sensors are becoming popular for rapid *in situ* measurement of C emissions (Kominami and Takami, 2004). The closed chamber method is the simplest and most commonly used technique and consists in measuring the continuously accumulating C emissions from the soil inside a static or non-static chamber (Fig. 17.9). A closed chamber consists of a bottom chamber and a lid where the bottom portion is inserted into the soil. The chamber covered with the lid allows the accumulation of gas inside the chamber during sample collection over a specific period of time (e.g., 0 to 60 min). The daily flux of C in g m^{-2} day^{-1} is computed as

Fig. 17.9 The closed chamber method consists of a gas sampling chamber made of PVC with a bottom section (30 cm long × 15 cm diameter) inserted into the ground, and a lid equipped with a gas sampling port (Photo by H. Blanco). Air samples withdrawn from the chamber are stored in evacuated vials for the soil gas (e.g., CO_2, CH_4) analyses

$$Flux = \left(\frac{\Delta C}{\Delta t}\right)\left(\frac{V}{A}\right)k \tag{17.3}$$

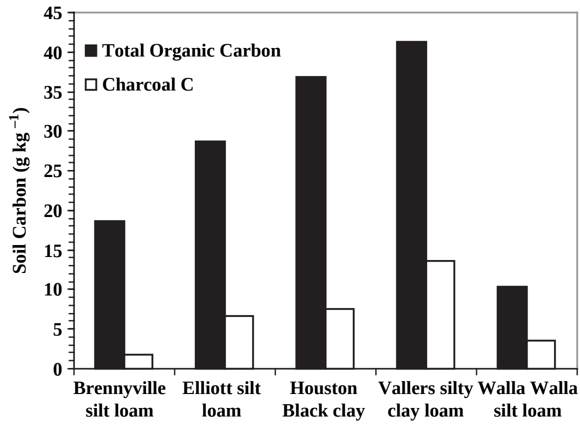
where $\frac{\Delta C}{\Delta t}$ is rate of gas accumulation inside the chamber (g cm^{-3} min^{-1}), V is chamber volume (cm^3), A is chamber area including the bottom and headspace (m^2), and k is time conversion factor (Jacinthe et al., 2002).

17.19 Biochar

An important component of total soil C is charcoal, also known as black C or biochar. Charcoal is an impure form of C and is mostly found as recalcitrant C. Black C consist of aromatic C and can make up to 10 to 45% of the total soil C in some environments (Fig. 17.10). Charcoal results from slow smoldering or incomplete burning of biomass from woodlands, shrublands, grassland, and agricultural crops (e.g., stalks). The frequent and extensive fires in historical times caused formation of relic charcoal which persist for long periods of time. Unlike organic C, charcoal is not biologically active and is thus a potential means of long-term C sequestration. When the total soil C concentration decreases due to the rapid turnover of soil organic C, the charcoal, which is a resistant portion of the total C, increases in terms of percentage of the total C. High concentration of charcoal C can cause the overestimation of the amount of C. Separation techniques of charcoal C from the organic fraction must be used.

Burying charcoal is a viable option for storing C in soil while improving crop production. In Brazil, a number of human-made sites called "black earth" (terra petra

Fig. 17.10 Distribution of total soil organic C and charcoal C for five principal soils in the USA (After Skjemstad et al., 2002)



in portuguese) exist along the Amazon River region where buried charcoal below the soil surface thousands of years ago restored degraded ecosystems into highly productive lands while enhancing a long-term C storage (Lehmann et al., 2003). Slash-and-burn agriculture produces large amounts of greenhouse gases. Replacing slash-and-burn with slash-and-char, a process in which charcoal is produced by slow burning of biomass covered with soil and straw under limited supply of oxygen, can annually reduce net emissions of C while improving soil fertility and thus reducing the use of fertilizers.

17.20 Modeling Soil Carbon Dynamics

Measurement of erosion-induced changes in C pools for each soil is expensive. Modeling is a useful tool for estimating management impacts on soil C dynamics. Carbon models can estimate gains in C storage under various management scenarios. Among the common C models are EPIC, Century, APEX, Ecosys, and CQESTER. The ability of models to simulate C dynamics is a function of how well the input parameters reflect the soil management. Properties of soils must be measured over time and space for a detailed modeling of C storage and fluxes in no-till soils because soil processes and properties vary within and among soils. The C distribution in soils due to tillage erosion can also be modeled. The SPEROS-C is one of the models to estimate the effect of tillage erosion on C distribution over a hillslope. This model simulates lateral and vertical translocations of C with soil. Pedotransfer functions (PTFs) and principal component analysis (PCA) are useful tools for studying site-specific relationships between C concentrations and other soil properties. Data on soil properties from long-term experiments are important input parameters for process-based models and PTFs to predict the ability of improved management practices to enhance C concentration.

17.21 Soil Conservation and Carbon Credits

Increase in soil organic C pool through the use of soil conservation practices not only reduces soil erosion but is also a tradable commodity and has an economic value. Trading C credits through C storage can provide additional income to farmers. Farmers adopting conservation practices could sell the generated C units to industries or companies that are currently emitting C. The more crop residues are returned to soil or more permanent vegetation is grown, the greater the amount of C stored, and the more the opportunities for trading C units. Residue mulching and growing vegetation reduces oxidation of soil organic matter and release of C. These conservation practices also reduce C mineralization by reducing soil erosion, which exposes C to rapid mineralization. Although the trading system and price for C credits are still under development, regional C markets are emerging rapidly. Thus, it is conceivable that trading C credits across the globe will become important once price regulations and trading policies are established which will entice farmers to adopt new climate change mitigation technologies. Thus, C sequestration should generate economic revenues to farmers if the gains in C under no-till and other improved soil conservation technologies are to be permanent. Carbon sequestration could increase net farm income, which may stimulate the desired changes in land use and tillage management systems.

Monitoring and assessment of impacts of conservation technologies on the rate of C sequestration are essential to the development of C credit trading systems. Ecosystem C budget for different regions must be estimated based on above-ground biomass (e.g., land cover type, detritus material, canopy cover, crop residues) and below-ground ground biomass (e.g., root biomass, root respiration), crop yields, rates of decomposition of organic matter, and measured C fluxes. The C budgets developed at smaller or local scales can be eventually applied to regional and global scales for trading C credits. Carbon budget models are used for estimating regional annual C fluxes. For example, C for the crops is computed as

$$C_{gr} = W_g f_c Y \tag{17.4}$$

$$C_p = \frac{C_{gr} W_g}{HI} \tag{17.5}$$

where C_{gr} is C in the grain, W_g is grain moisture content, f_c is fraction of C in the grain, and Y is crop yield, C_p is below-ground biomass, and HI is harvest index (Hollinger et al., 2005).

Gains and losses in soil C are estimated as

$$\begin{aligned}\Delta \text{C Storage} &= \text{Input} - \text{Output} = \text{Crop residue} + \text{Manure} + \text{Runon} \\ &\quad + \text{Aerial Deposition} - \text{Mineralization} - \text{Erosion} - \text{Leaching}\end{aligned}$$

The soil organic C sequestration rates, inputs levels (e.g., herbicides), field operations (e.g., equipment, labor), and crop yields must be quantified to support the development of C credits for all management systems. Soil organic C concentration prior to and following establishment of conservation technologies must be determined to estimate net C storage. Data on gains in C upon making changes in land use and no-till management are converted to monetary value. For example, no-till systems having higher net C sequestration rates and higher net economic returns (e.g., grain and biomass yields) may not require C credit payments to stimulate the adoption of no-till practices. In contrast, ecosystems with significant gains in net C sequestration rates but reductions in crop yields upon introduction of no-till systems may require C credit payments to entice producers to adopt C-sequestering methods. Net returns, net C sequestered, and production costs will be used to estimate C credit values as

$$C_{value} = \frac{NR_a - NR_b}{C_a - C_b} \tag{17.6}$$

where C_{value} is C credit value, NR_a and NR_b are net returns, and C_a and C_b are C sequestration rates for management *a* and *b*, respectively (Sandor and Skees, 1999).

Summary

Soil organic C is an important component of the terrestrial C pool. More C is stored in the soil compared to either in the terrestrial biomass or in the atmosphere. Excessive plowing and burning and removal of crop residues deplete the soil organic C concentration and increase the atmospheric C concentration. Soil organic C stabilizes the soil against soil erosion and increases or maintains crop production. Accumulation of C in the soil not only purifies the atmosphere but also maintain water sources clean by absorbing and filtering point- and non-point-source pollutants.

Soil erosion by water, wind, and tillage is one of the pathways of C loss. Water erosion in sloping lands and wind erosion in flat landscapes contribute to losses of C. Soil aggregate disintegration and dispersion, preferential removal of C, redistribution of eroded C, and mineralization of eroded C are some of the mechanisms by which the soil C is lost. The eroded C consists mostly of labile organic fractions which are thus prone to rapid mineralization. Some of the eroded C in depositional areas is lost in the form of C emissions and some is buried, promoting long-term C sequestration. The fate of eroded C is complex. Eroded C can be oxidized at the eroding site, during transport, depositional zones, and aquatic systems. Thus, soil erosion is probably a source rather than sink of C.

The soil C lost through anthropogenic activities must be brought back to where it belongs. Soil conservation practices are strategies to store C, reduce soil erosion, and improve crop productivity. By leaving crop residues on the soil surface and reducing soil disturbance, no-till farming is one of the top innovative practices that can promote C storage. It promotes soil aggregation, which is essential to store and protect organic materials from rapid decomposition. No-till benefits to increasing C sequestration are, however, site-specific. Increases in soil C in no-till are mostly confined to the soil surface where residues are concentrated. The total C pool between no-till and plow tillage practices for the whole soil profile may not differ.

Crop rotations, intensive cropping systems, cover crops, crop residues, manure application, agroforestry, and high-biomass producing bioenergy crops are practices that increase soil C while conserving soil and water. Reclaiming degraded lands with growing vegetation is a potential alternative for increasing C pools in terrestrial systems. The stored C is tradable and has an economic value. The more C is stored in the soil, the greater the opportunities for trading C units. The C trading system is developing and it is expected to become important to conserve soil and manage C.

New methods are emerging for rapid measurement of soil organic C concentration under *in situ* conditions based on atomic emission spectroscopy, neutron scattering, infrared reflectance spectroscopy, and remote sensing. These methods are being calibrated for different soils and refined in their resolution to estimate small changes in C concentration. Modeling is a useful companion to direct measurement techniques to estimate soil C storage and extrapolate information across a large geographic spectrum under different tillage and cropping management scenarios.

Study Questions

1. Explain in detail the mechanisms by which soil organic C reduces soil erosion.
2. Discuss erosion models that incorporate soil organic matter as input to erosion modeling.
3. Discuss the specific processes by which soil plowing reduces soil organic C concentration.
4. Explain differences in the mechanisms of C removal among water, wind, and tillage erosion. Provide estimates of the amount of C transported by each component.
5. Discuss the fate of C transported by erosion and provide quantitative estimates of the main pathways of fate.
6. List pros and cons of the new methods of C analyses with the conventional methods.
7. Discuss the mechanisms by which no-till farming would store more C in the soil.
8. A soil has a bulk density of 1.3 Mg m^{-3} and 2.5% of organic C concentration at a soil depth of 10 cm. Estimate the amount of C stored in Mg ha^{-1}, Mg km^{-2}, and g kg^{-1}. How would one estimate the C pool on a volume basis if data on bulk density are unavailable.?
9. Calculate the C sequestration rate for a soil under alley cropping systems with 2-m wide hedgerows and 8-m wide alleys for an 80 m wide $\times$122 m long field. Soil bulk density is 1.45 Mg m^{-3} within alleys and 1.1 Mg m^{-3} within hedgerows, while C concentration is 25 g kg^{-1} within alleys and 36 g kg^{-1} within hedgerows.

10. Calculate the C sequestration rate for the same soil in Problem 9 but under silvopasture. Trees within silvopasture are established in four 8-m wide rows and 30-m apart. Soil bulk density is $0.90\,\mathrm{Mg\,m^{-3}}$ under the trees and $1.1\,\mathrm{Mg\,m^{-3}}$ under pasture, while C concentration is $65\,\mathrm{g\,kg^{-1}}$ under trees and $31\,\mathrm{g\,kg^{-1}}$ under pasture.
11. Concentrations of CO_2 samples were determined using the $0.15\,\mathrm{m}$ diameter $\times 0.30\,\mathrm{m}$ high closed soil chamber technique at random points across a $100 \times 200\,\mathrm{m}$ long field under forest farming. Calculate the emissions of CO_2 in $\mathrm{g\,ha^{-1}\,day^{-1}}$ if the average concentration of CO_2 in ppm was 350.

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Chapter 18
Erosion Control and Soil Quality

Soil is a living, complex, and three-dimensional body, which undergoes continuous and dynamic changes. Rapid changes in energy fluxes and organic matter decomposition reflect the dynamic nature of the soil. The need for a greater understanding of implications of different land use and management scenarios on soil function resulted in the emergence of a conceptual paradigm in late 1970's known as *soil quality* (Warkentin and Fletcher, 1977). This concept has received greater attention since 1990's (Karlen et al., 1990; Larson and Pierce, 1991). It stemmed from an innovative perspective that attempts to define how a soil functions and what measures or management practices maintain and improve a soil for a specific use. Soil quality concept has attained importance for merging traditional concepts of soil taxonomy with management and conservation to address growing concerns about the depletion of natural resources, non-point source pollution, and the projected global climate change. Present and future needs of food production and environmental protection depend on how the soil responds to external and internal stresses. Soil attributes in interaction with science-based agricultural inputs determine soil productivity. Introduction of soil quality concept represents an innovative paradigm in soil science research.

18.1 Definitions of Soil Quality

Conceptual definitions and assessment tools of soil quality are still evolving. At this point, definitions of soil quality vary depending on the views and the background of individuals (Table 18.1). Early definitions associated soil quality with productivity, which is the capacity of a soil to produce a plant or sequence of plants under a given management system. Contemporary definitions equate soil quality with sustainability, environmental quality, and global climate change in addition to productivity. The term "fitness for use" has been proposed as a simple working definition of soil quality (Larson and Pierce, 1994). In this textbook, soil quality is defined as the soil's intrinsic ability to perform a specified function. It refers to the capacity of a soil to buffer anthropogenic perturbations, maintain productivity, moderate pollutants, protect watersheds, and improve water and air quality. This is certainly a complex and multifunctional concept.

H. Blanco, R. Lal, *Principles of Soil Conservation and Management*,

Table 18.1 Some modern soil quality concepts

Definitions of soil quality
• [1]The capacity of a soil to function within ecosystem boundaries to sustain biological productivity, maintain environmental quality, and promote plant and animal health.
• [2]Capacity of a soil to function, within ecosystem and land use boundaries, to sustain biological productivity, to maintain environmental quality, and promote plant, animal and human health.
• [3]The capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, to maintain or enhance water and air quality, and support human health and habitation.
• [4]The soil's fitness to support crop growth without resulting in soil degradation or other otherwise harming the environment.
• [5]The capacity of a soil to function within its ecosystem boundaries and interact positively with the environment external to that ecosystem.
• [6]Inherent attributes of soil and characteristics and processes that determine the soil's capacity to produce economic goods and services and regulate the environments.
• [7]The ability of the soil (i) to accept, hold, and release nutrients and other chemical constituents; (ii) to accept, hold, and release water to plants and surface and groundwater recharge; (iii) to promote and sustain root growth; (iv) to maintain suitable soil biotic habitat; and (v) to respond to management and resist degradation.

[1]SSSA (1997), [2]Doran et al. (1996), [3]Karlen et al. (1997), [4]Acton and Gregorich (1995), [5]Larson and Pierce (1994), [6]Lal (1993), and [7]Larson and Pierce (1991).

18.2 Divergences in Conceptual Definitions and Assessment Approaches

There is a growing debate about the objectivity and scientific basis of soil quality concept. While some view that soil quality concept offers a new approach for addressing agricultural productivity and environmental protection (Karlen et al., 2006), others believe that the concept of soil quality is highly subjective, value-laden, and ill-defined (Letey et al., 2003; Sojka et al., 2003). This debate has somewhat distracted from the focus on one common goal, which is the conservation and appropriate management of soil and water resources. Some concerns are that adopting prematurely the soil quality paradigm without having the support of a well-founded scientific basis may "lead to advocating a system as an end unto itself" (Sojka and Upchurch, 1999).

Soil quality research is still in its infancy. The concepts and parameters for soil quality assessment require further refinement. At present, standards and guidelines for a rigorous scientific evaluation of soil quality are unavailable. Some researchers have used the terms "soil quality" and "soil health" interchangeably, creating further controversies and confusion. Soil quality is not directly quantifiable, which makes the concept somewhat elusive with no clear goal of judgment. Unlike pure substances such as water and air, a soil is a complex three-phase system because it consists of a mixture of water, air, organic and inorganic materials in addition to live organisms.

Quantitative standards, thresholds values of soil properties, and practical guidelines of soil quality evaluation must be developed and tested across a wide range of soils and management conditions. Physical, chemical, and biological properties of the soil are measurable but are site and purpose specific. Some of the current soil quality assessment guidelines are too:

- Simplistic (e.g. scoreboard approach)
- Subjective (e.g., interviewee bias)
- Non-technical and qualitative (e.g. use of soil aroma as an indicator)
- Reductionist (e.g., few soil properties)

These approaches raise concerns and conflicting views about the scientific basis of soil quality concept. Simplistic approaches undermine the scientific method of the discipline of soil science, which envisions to parallel advanced scientific fields of physics and chemistry (Sojka and Upchurch, 1999). Moreover, some of the current scoring functions of soil quality indicators emphasize only on positive weights and not negative impacts. For example, indicators such as high organic matter concentration and earthworm population are weighted positively for a “good” soil in regards to crop production because of their essentiality to nutrient cycling and improved macroporosity, water infiltration, and drainage. The same indicators can, however, have negative impacts on water quality. Because of its high pesticide sorption capacity, elevated organic matter input can increase use of pesticides with the subsequent negative impact on water quality. Similarly, earthworm population is rated as highly beneficial to soil quality somewhat disregarding that preferential or by-pass flow of chemicals through earthworm burrows can lead to groundwater pollution in soils with abundance of earthworms (Shuster et al., 2003).

18.3 New Perspective

Theoretical concepts and assessment techniques of soil quality must be refined. The scientific debate about the soil quality concept is vital to the refinement of new ideas and knowledge. The existing diverse views on the methods of assessment and interpretations of soil quality provide an opportunity to revisit and improve conceptual definitions and assessment approaches. New concepts in science have always emerged from a paradigm of controversies and disagreements, which are essential to the development of solid and sound concepts.

The controversies in soil quality are mostly in regards to its definition and assessment methods rather than to its importance. Most soil scientists recognize its importance, and support the need of developing sound approaches to address the seriousness of soil degradation, environmental quality deterioration, and human pressure on the limited soil resources. What some scientists have argued is the premature use of the soil quality concept without having standards or definite scientific criteria of evaluation (Sojka and Upchurch, 1999).

Renewed and conciliatory communications in soil quality concepts and methods of assessment are warranted to strengthen the common long-term objectives of soil and water conservation. Opposing views must propose viable alternatives to characterize soil from a scientific perspective. Development of effective strategies to better conserve and manage soils is preferred over controversies. Because current concepts and approaches of soil science have not entirely addressed ongoing problems of soil degradation and non-point source pollution, development and reinforcement of new approaches, such as soil quality concept, are plausible goals as alternative for a better management of soils and protection of environment. The complexity associated with soil quality concept and assessment techniques must not be a deterrent for a continued research on soil quality for developing a robust assessment approach.

The indicators of soil quality must be tested against scientific soundness, applicability across a range of soil ecosystems and management objectives, and reliability (Bremer and Ellert, 2004). Indicators must be simple to be understood by users, yet complex enough to account for all the interactive factors defining soil function. A relevant indicator of soil quality is the one that soil scientist can quantify but farmers can also relate to and assess by using standard approaches. To make the soil quality more adaptable to a wide range of management and soil systems across the world, adoption of a broader perspective of soil quality research accompanied by standard tools and methods is necessary.

18.4 Soil Quality Paradigm and its Importance

Emergence of the soil quality concept is particularly important at a time when increase in land degradation, non-point source pollution, and emissions of greenhouse gases into the atmosphere are among major global concerns. The modern term "soil quality" was introduced first for soils in temperate zones, but it is of equal interest in its practicality and adoption worldwide, and especially in developing countries where deforestation, food insecurity, poverty, and soil degradation are main concerns. Unlike in temperate zones where non-point source pollution is one of the main environmental problems, in tropical regions, such as is the case in Africa, an acute problem is nutrient depletion which compounded with soil degradation has stagnated agricultural production and biodiversity.

The evolving concepts of soil quality complement soil science research and make the traditional concepts of soil management more practical. The development of new concepts of soil management is crucial to understanding soil response and use in a time when human activity is exerting increased negative pressures on the land. Soil quality is a potential educational tool for conservationists, extensionists, land managers, and farmers for assessing management impacts on soil resources. It can complement other large-scale evaluation systems including land capability classification, integrated natural resource management framework, and fertility capability soil classification system used mainly in the tropics (Sanchez et al., 2003).

Soil quality attempts to integrate inherent and dynamic soil properties as influenced by management based on threshold values of soil properties. Using this approach, data on soil properties are converted into simple but useful indices based on the present knowledge and experience to more effectively serve the land users and policy makers than traditional approaches. It is a management tool that can elucidate how a specific practice influences soil behavior or how the soil responds to management over time. With the development of appropriate soil quality indices based on sound scientific principles, soil quality concept can change the way soil is managed. It can help in evaluating early signs of soil degradation as well as designing measures of prevention and reclamation. The basic understanding of soil properties and processes through practical and theoretical indexes is a new approach to better manage the soil resources.

Soil is a fundamental resource that performs multiple and simultaneous ecosystem functions. Specific functions are the following:

- Agronomic productivity
- Nutrient recycling
- Storage and purification of water and air
- Energy exchange
- Wildlife habitat and biodiversity
- Reduction and moderation of greenhouse gas fluxes
- Storage, sequestration, and recycling of C and other elements

A soil with good quality not only must produce high-quality crops and forages but also must protect the environment. Transport and fate of water pollutants and reduction of emissions of greenhouse gases depend on the soil's intrinsic ability to filter/degrade pollutants and sequester C, respectively. These contrasting but simultaneous functions of soil make the development of standard indicators difficult.

18.5 Indicators of Soil Quality

Because soil quality can not be measured directly, it is estimated from "indicators" that affect one or more simultaneous functions. A soil quality indicator is a measurable soil property, which influences the capacity of a soil to function for a specific purpose (Karlen et al., 1990). Soil physical, chemical, and biological attributes that are sensitive to change in land use and management and vary over time and space are indicators of soil quality (Table 18.2). These indicators are not independent but strongly interrelated. The dynamic nature of soil quality requires a detailed monitoring and selection of the indicators for different objectives.

Rapid tests of soil quality in agricultural soils have often relied on changes in macronutrient levels. Expanded methods have integrated information on nutrient levels with soil mineralogical composition, texture, structural properties, and biochemical and microbial characteristics, which rapidly change with erosion and management. Soil quality assessment begins with the selection of minimum data sets of soil properties or indicators from field and laboratory data.

Table 18.2 Indicators of soil physical quality [After Lal (1997) and Bremer and Ellert (2004)]

Soil profile characteristics	Soil structure	Dynamics of water, air, and heat flux
• Profile depth • Root zone depth • Horizonation • Sand, silt, and clay contents • Clay mineralogy	• Crusting • Cone index • Shear strength • Bulk density • Aggregate size and stability • Macro- and micro-porosity • Pore-size distribution • Aeration	• Drainage • Leaching • Water infiltration • Hydraulic conductivity • Plant available water capacity • Air permeability • Soil temperature • Air filled-porosity

18.5.1 Soil Physical Quality

Change in soil depth due to erosion is a major indicator of soil quality as it affects nutrient and water storage and crop production. Near-surface (0–10 cm) strength parameters indicate the degree of soil compactness. Likewise, changes in aggregate size, strength, and stability are important determinants of soil structural quality and the ability of soil to resist erosional processes. Aggregate stability, for example, portrays the soil response to detachment under raindrop and runoff forces. Information on water infiltration rate is also a critical indicator because it determines the partitioning of rainwater into runoff and infiltration. Soil quality changes over time and hardly remains static. Periodic monitoring of dynamic soil physical properties is important to determining the change in soil surface quality over time.

18.5.2 Soil Chemical and Biological Quality

Change in soil organic matter concentration is a common indicator of soil chemical and biological quality because of its significant effects on biological activity, CEC, pH, and nutrient levels across a wide range of soils (Brejda et al., 2000) (Table 18.3). Presence of earthworms is another important indicator of a desirable soil quality. Earthworms ingest crop and process residues, release essential nutrients, favor microbial processes, soil aggregation, and water and air movement through the soil profile. Changes in microbial population and activity due to shift in management (e.g., no-till, residue return) are also vital indicators of soil quality.

18.5.3 Macro- and Micro-Scale Soil Attributes

A comprehensive assessment of soil quality requires an integrated approach. Soil quality is affected by *management* in interaction with *soil type, topography, vegetation*, and *climate*. Inherent differences in soil forming factors and anthropogenic

Table 18.3 Indicators of soil chemical and biological quality [After Lal (1997) and Bremer and Ellert (2004)]

Soil chemical characteristics	Soil biological characteristics
• Soil organic matter content • Nutrient content and availability • pH • Electrical conductivity • Sodium adsorption ratio • CO_2 concentrations • Cation exchange capacity (Ca^{2+}, Mg^{2+}, K^+, Na^+) • Toxicity (Al^{3+}, Mn^{2+}) of elements • Acid drainage (e.g., heavy metals)	• Soil respiration • Ergosterol concentrations • Macro-organisms (e.g., earthworms) • Micro-organisms (e.g., nematodes, protozoa) • Macroflora (e.g., mosses) • Microflora (e.g., bacteria, fungi, algae)

interventions determine differences in soil quality. Soil quality is the product of an integrated influence of intrinsic and extrinsic factors. It can not be fully determined unless all major factors that affect soil formation and land attributes are considered. As the soil series and individual soils differ within and among fields, watersheds, and regional levels, so does soil quality among soil types and landscape units.

External attributes of the landscape (e.g., soil slope, surface geomorphology) are closely related to internal soil attributes (e.g., physical properties) which influence soil quality. For example, changes in soil slope and alterations in landforms due to removal and deposition of soil caused by soil erosion are the dynamic indicators of soil quality. Definition of soil quality warrants the assessment of all macro- (e.g., properties of the bulk soil) and micro-scale (e.g., aggregate properties) soil attributes. In well-aggregated soils, properties of discrete aggregates can be more responsive to management than those of the bulk soil and are perhaps better indicators of soil quality (Blanco-Canqui et al., 2006b). To date, research on indicator selection has mostly emphasized on properties of bulk soil rather than on individual aggregate properties, which control the behavior of the whole soil. Some of the aggregate properties include strength, stability, density, water retention capacity, sorptivity, wettability, and saturated and unsaturated water flow.

18.5.4 Interaction Among Soil Quality Indicators

The physical, chemical, and biological indicators are interdependent and interact to determine the quality of a soil. For example, most soil structural properties are significantly correlated with organic matter concentration. Thus, soil quality indexing must emerge from a unique balance and interaction of all soil properties and processes. Correlated soil properties do not respond independently to management change but in interaction with other properties (Brejda et al., 2000). Single soil properties used as indicators do not account for the many dynamic interacting factors and processes. Soil properties that respond to more than one function are preferable to capture most of the variability and address the multiple soil functions for one entire soil series or at regional scales, depending on the complexity of the soils. Multiple soil series within local and regional scales and high variability in soil properties even within a single soil series hinder the selection of a unique set of indicators.

18.6 Soil Quality Index

No single soil quality index (SQI) can be applicable to all soils. A refined technique must reflect the complexity of the soil system, consider the specific land use, and be based on specific standards for each soil, landscape position, crop, and management. Specific indexing examples of soil quality must be developed for representative soils. A SQI must not be based only on a few, often arbitrarily, selected soil properties without accounting the interactive nature of complex and numerous soil physical, chemical, and biological properties. For example, the use of single indicators, such as organic matter concentration, is not sufficient to evaluate response of soils to changes particularly in soils with reduced effective depth. A proper weighting of soil quality indicators for serving various simultaneous functions (e.g., crop production, water quality) must be developed. Indexing should consist of various elements. Mathematical basis along the lines proposed by Larson and Pierce (1991) shown in Eq. (18.1) must be developed

$$\frac{dQ}{dt} = f\frac{\left(\frac{q_{it}-q_{it_o}}{q_{it_o}} \cdots\cdots \frac{q_{nt}-q_{nt_o}}{q_{nt_o}}\right)}{dt} \tag{18.1}$$

where Q is soil quality, t is initial time, and t_o is time when soil properties or indicators are measured. Although Eq. (18.1) is not a physically based model, further research and knowledge are required to develop theoretical models of this kind for soil quality evaluation.

18.7 Assessment Tools

Selection of representative indicators requires the use of advanced statistical tools (e.g., multivariate analyses) and modeling approaches to account for the significant correlations or complex interactions among dynamics and static soil properties. Multivariate statistical and canonical discriminate analyses are tools to identify relevant parameters for soil quality assessment (Giuffré et al., 2006). Principal component analysis (PCA), a multivariate statistical approach, is a common tool to select representative minimum data sets of soil properties (Table 18.4). The PCA is also used in crop yield predictions based on critical soil properties. It uses linear combinations of soil properties to determine the maximum variance within a data set consisting of a large number of soil properties. It groups soil variables in one or various principal components (PCs) according to the importance and affinity of variables while reducing the dimension of the original data set without losing the overall information of the data set. The PCA reveals important trends in

Table 18.4 Some common soil quality indicators identified in selected studies

Method of identification	Soil quality indicators	Management
PCA[1]	Cone index, shear strength, soil water content, soil temperature, mean weight diameter and organic matter content	Crop production
PCA and discriminant analyses[2]	Organic matter concentration, hydraulic conductivity, and soil strength.	Reclamation of degraded soils
Arcview GIS, GPS receiver, and PCA[3]	Topographic attributes (slope and elevation) and soil attributes (very fine sand, base saturation, pH, clay content)	Crop production
PCA[4]	Organic matter concentration, total N, microbial biomass C and N, Exchangeable K, P, extractable Fe, Mn, and Zn.	Crop production
Multivariate approach[5]	Particulate organic matter and organic matter -related biophysical soil properties	Crop production
PCA[6]	Soil electrical conductivity and soluble Mg and Na	Wetland management

[1]Blanco et al. (2006a), [2]Xu et al. (2006), [3]Jiang and Thelen (2004), [4]Andrews et al. (2002), [5]Wander et al. (2002), and [6]Richardson and Bigler (1984).

relationships and identifies parameters of interest within each PC. The condensed minimum data set by PCA is interpreted directly or subjected to further analyses for assessing soil quality (Andrews et al., 2002). Simple correlations and pedotransfer functions are potential tools to relate independent to dependent variables. The pedotransfer functions have been used to predict a range of dynamic soil properties (e.g., hydraulic properties) and crop yields from readily available soil datasets in relation to soil quality. Simulation models (e.g., hydrological models) are also useful to soil quality evaluation in that they model highly complex indicators (Wösten, 1997).

Over the last 15 yr, research on soil quality has resulted in numerous workshops, reports, and extension guidelines and quality assessment approaches designed to understand and expand the soil quality concept (Karlen et al., 1990; Doran and Parkin, 1994; Lal, 1994; Andrews et al., 2004). What started as a qualitative, subjective, and simplistic term is gradually being refined and incorporated into soil management decision support systems. Methodologies are being widely assessed under different ecosystems. Because of its site- and purpose-specificity, the development of scoring functions and indexes has been dictated by the intended use of the soil.

18.7.1 Farmer-Based Soil Quality Assessment Approach

Early soil quality indexing was based mainly on simple scoring functions and qualitative approximations. Soil quality assessment has often involved farmer participation. The Wisconsin Soil Health Scoreboard (WSHS) (Romig et al., 1995) and the

Illinois Soil Quality Initiative (ISQI) (Wander et al., 2002) are examples of evaluations of soil quality based on farmer-based surveys. The WSHS uses scores ranging from 0 to 4 where 0 is for the least favorable values of soil property and 4 is for the most optimum values. The WSHS integrates soil surface characteristics (e.g., surface residue cover, degree of erosion, ease of tillage, surface crusting and cracking), soil profile attributes (e.g., topsoil depth, drainage, depth of A horizon), and soil properties (e.g., earthworm population, soil structure development, soil color, compaction level, water infiltration, water retention, soil fertility level, degree of decomposition of organic residues, hardness, soil texture, aeration, biological activity), and soil qualitative indices (e.g., feel, smell). The ISQI scores selected soil properties associated with nutrient content, water relations, and root growth zone using scores of 0.1, 0.5, 0.75, and 1 for low, moderate, intermediate, and high soil quality, respectively.

18.7.2 Soil Test Kits

Soil quality has also been assessed using inexpensive and commercially available test kits (Sarrantonio et al., 1996; USDA-NRCS, 2003). The test kits are designed for quick assessment of soil properties under field conditions, and accompanying guidelines are used to interpret the results and estimate the soil and water quality for a specified land. Field test kits may produce results comparable to those from laboratory, depending on the soil property, sampling time, soil disturbance, soil depth, and number of replications (Liebig et al., 1996). The use of test kits is particularly popular among extension workers, educators, and farmers because of its simplicity. The main soil properties measured with the test kits include soil respiration, water infiltration, bulk density, EC, pH, soil nitrate content, aggregate stability, slaking, and earthworm population. These soil properties are combined with visual observations of soil structure, root biomass, topsoil depth, degree of erosion, compaction, and soil profile texture to estimate the prevailing soil quality. The test kits are promising tools for quick and point measurement of selected soil properties although additional research is needed to validate the accuracy of test kits across a range of different soil and management conditions.

18.7.3 The Soil Management Assessment Framework

A recent advance in soil quality indexing is the development of soil management assessment framework (SMAF) proposed by Andrews et al. (2004). The SMAF is the result of the ongoing research on soil quality indexing, and it is thus still under development. It is a computer-based program with specific codes and algorithms in Excel spreadsheets. Case studies have shown that SMAF is promising to examine and monitor soil quality across different soils, climates, land uses, and management conditions (Andrews et al., 2004). The overall approach of the SMAF involves defining the management goals, obtaining data on soil properties, and selecting, interpreting, and scoring indicators to compute the SQI. The three main steps of SMAF are selection, interpretation, and integration of indicators (Andrews et al., 2004). Based on the estimated scores, a single numeric value, known as SQI, is computed either using Eq. (18.2) (Andrews et al., 2002) or Eq. (18.3) (Andrews et al., 2004) as follows:

$$\mathrm{SQI} = \sum_{i=1}^{n} \mathrm{Wi} \times \mathrm{Si} \tag{18.2}$$

$$\mathrm{SQI} = \frac{\sum_{i=1}^{n} \mathrm{Si}}{n} \times 10 \tag{18.3}$$

where *Wi* is weighting factor using principal component analysis and *Si* is score of soil quality obtained from the scoring curves. These SQI values are used to: (1) monitor changes in soil quality over time for the same soil and (2) compare soils within the same management practice. The SQI simplifies the parameters by providing scores and is a promising assessment tool to reorient soil management and implement corrective measures when necessary.

18.8 Soil Quality and Erosion Relationships

Soil quality and erosion are strongly interrelated. Magnitude of erosion effects on soil quality depends on land use and tillage management (Lal et al., 1999). Erosion directly alters the indicators of soil quality. It alters the soil profile depth and soil physical, chemical, and biological properties as follows: For example, soil organic matter content decreases with increase in removal of topsoil.

18.8.1 Soil Erosion and Profile Depth

Thickness of the topsoil and the total soil profile depth are indicators of soil quality for crop production. Thickness of topsoil horizons decreases linearly with increase in soil erosion, decreasing the total depth of soil profile. Soil erosion truncates the upper horizons and exposes subsurface horizons which result in immediate losses of organic matter and nutrients, and deterioration of near-surface soil physical properties. Severe erosion can completely remove the Ap horizons. The adverse effects of erosion on soils with shallow surface layer may be irreversible. Surface soil is a medium that partitions the rainfall into different hydrologic components and controls surface runoff. Thus, losses of topsoil diminish the soils ability to retain water and nutrients. Exposed subsurface horizons often have higher runoff and soil erosion risks because of reduced soil structural development and low organic matter concentrations. Soil aggregates bound by organic matter are more porous and stable than those bound by clay. Erosion reduces the effective soil depth, which is the depth between soil surface and the root-restrictive subsoil horizons. Most of the important soil processes occur within the effective rooting depth, and thus any reduction in its depth has negative effects not only on soil properties but most importantly in crop production.

18.8.2 Soil Physical Properties

Erosion exposes subsurface horizons with different properties from the uneroded topsoil (Table 18.5). The exposed horizons have adverse and often fragile structural properties. Eroded soils are prone to surface sealing and crust formation under raindrop impacts, reducing the water infiltration and affecting soil structural formation. Exposed subsurface horizons are also lighter and have higher clay content especially in soils with Bt horizons, and are prone to cracking. Soil erosion alters soil texture by exposing subsurface layers of different texture and by preferentially removing fine soil particles. Erosion is a selective process in that small primary and secondary particles along organic matter are more rapidly transported by runoff water. Bulk density and penetration resistance increase with increase in topsoil removal.

Table 18.5 Impact of soil erosion on soil properties across various soils

Soil	Soil property	Slightly eroded or uneroded	Severely eroded
Silt loam[1]	Bulk density (Mg m^{-3})	1.3	1.4
	Water content (g kg^{-1})	325	215
	Mean weight diameter (mm)	2.7	1.3
	Soil organic C (g kg^{-1})	20	15
Fine sandy loam[2]	Bulk density (Mg m^{-3})	1.4	1.5
	Cation exchange capacity (meq 100 g^{-1})	7	10
	Soil organic carbon (g kg^{-1})	11	7
	Clay content (g kg^{-1})	80	160
Sandy loam[3]	Infiltration rate (mm h^{-1})	62	4
	Depth to Bt horizon (m)	0.7	0.1
	Soil color	7.5YR 4/4	2.5YR 4/6

[1]Lal et al. (2000), [2]Mokma and Sietz (1992), and [3]Radcliffe et al. (1990).

Soil erosion often increases bulk density and decreases plant available water, water infiltration rate, and hydraulic conductivity because of decrease in macroporosity and increase in massive structure in lower horizons (Table 18.5). Decrease in saturated hydraulic conductivity increases soil loss and runoff (Blanco-Canqui et al., 2004). Magnitude of changes in soil properties with erosion varies among soils. In soils with high organic matter concentrations and deep horizons, moderate soil losses may not significantly change soil properties. Changes in soil properties by erosion are typically slow and may pass unnoticed until the system reaches a severely degraded or an irreversible stage.

18.8.3 Soil Chemical and Biological Properties

Erosion by water and wind not only alters soil physical properties but also changes soil chemical properties. Loss of organic matter by erosion is the major cause of soil degradation because it is vital to sustain desirable soil physical, chemical, and biological properties of the soil. The organic matter concentration of the surface layers decreases linearly with increase in soil erosion in sloping fields. The organic matter lost is intermixed with eroded soil and bound to soil particles. Soil erosion displaces organic matter-enriched sediment off-site.

Soil erosion affects pH, composition of soil solution, CEC, and EC. In most soils, pH increases and CEC decreases with increase in soil loss. An eroded soil is chemically degraded because soil erosion alters essential chemical and biological processes including nutrient cycling, decomposition of organic matter, acidification, transformation, volatilization, and eutrophication. Essential nutrients and electrolytes normally bound to clay particles are transported off-site with eroded materials. Eroded soils are low in fertility and require large inputs of chemicals to compensate for the losses of nutrients. Changes in exchange complex (e.g., Ca^{++}, Mg^{++}, K^{+}) affect soil structural development and chemical nature of the whole soil.

18.9 Management of Soil Quality

Maintaining or improving soil quality through appropriate land use and soil management systems for a sustained crop production and environmental protection is a high priority. Management practices used for restoring degraded soils and improving soil resilience discussed in Chapters 15 and 16 are recommended practices for managing soil quality. Use of best management practices is a key to reducing soil erosion, and maintaining soil productivity. Reduction in tillage intensity and establishment of diversified crop rotation systems are key to improving the indicators of soil quality. In three different soils in the midwest USA, Karlen et al. (2006) reported that crop rotations which included 3 yr of forage crops had the highest soil quality index while continuous corn had the lowest. Crop yields are commonly lower under monocropping compared to those under diversified cropping systems.

Summary

Soil quality refers to the soil's intrinsic ability to perform a specific function. It depends on the specific use of soil. It has a widespread application in relation to management of natural resources, control of non-point source pollution, and amelioration of the projected global climate change. Soil quality is not only associated with productivity but also with environmental quality. A good soil is that which has a high productivity, buffers anthropogenic perturbations, filters and degrades pollutants, improves wildlife habitat, stores C, and reduces concentration of greenhouse gas emissions. Soil quality is a broad approach that attempts to integrate all inherent and dynamic soil properties and processes.

The theoretical definition and soil quality as well as its parameters of evaluation require further refinement. Current definitions are rather simplistic to fully capture the attributes of soils for a specific purpose. Standards and guidelines for a rigorous quantification of soil quality are unavailable. The complexity of the soil system, unlike pure substances such as water and air, limits the development of a solid framework of soil quality evaluation.

The soil quality concept is a promise to better manage soil if sensitive indicators that affect one or more simultaneous functions are identified. Soil physical, chemical, and biological attributes that are sensitive to change in land use and management are used as indicators of soil quality. Root zone depth, horizonation, surface conditions (e.g., sealing, crusting), compaction, aggregate stability, drainage, water infiltration, soil organic matter content, pH, CEC, and microbial biomass and activity. The interaction among these soil properties determines the quality of a soil.

A number of advanced statistical tools (e.g., multivariate analyses) and modeling approaches now are being used to identify indicators of soil quality for different soils and management scenarios. Field surveys and soil test kits are useful to assess soil quality. The soil management assessment framework is one of the recent advances in soil quality indexing based on computer codes and algorithms. Management of crop residues, adoption of reduced tillage and no-till systems, establishment of crop rotations, manure application, use of cover crops, nutrient management, and use of organic amendments are measures to manage soil quality.

Study Questions

1. Explain theoretical and practical differences between soil quality and soil health, outlining definitions and basic concepts.
2. Identify the most important indicators of soil quality based on available literature and discuss reasons for their selection.
3. Discuss how soil quality is related to soil degradation and resilience. Define each term.
4. Use data on soil properties from literature and identify sensitive indicators based on simple correlation coefficients.
5. Identify indicators in Prob. 4 using PCA and discrimination analyses.
6. The soil bulk density of a drastically eroded field was reduced from 1.40 to 1.1 $Mg\,m^{-3}$. Explain the reasons.
7. Compute the soil quality index for Prob. 5 using Eqs. (18.2) and (18.3).

8. Discuss how conservation buffers and agroforestry practices improve indicators of soil quality.
9. Explain relationships between soil quality and soil taxonomy.
10. Discuss the various soil forming factors and processes affecting soil function.

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Chapter 19
Soil Erosion and Food Security

Soil is the basis for crop production because about 99% of food is produced from the soil (Pimentel, 2000). Thus, food security depends directly on soil productivity. Accelerated soil erosion is among principal causes of the decrease in soil productivity and increase in risks of global food insecurity (Fig. 19.1). The magnitude of erosional impacts on ecosystem productivity and food security is, however, complex, variable, and soil specific. Crop production in regions with highly mechanized agriculture and large-scale farms coupled with the use of improved crop varieties, fertilizers, irrigation practices, and other advanced technological inputs has progressively increased since the 1960's, thereby masking the potential threat of erosion on food security. The increase in food production under intensive farming practices, however, must not be generalized across all ecoregions because it has not occurred in all regions, especially in Sub-Saharan Africa (FAO, 2006).

Fig. 19.1 A cropland affected by severe rill and ephemeral gully erosion (Courtesy USDA-NRCS)

Impoverished regions of the world with fragile soils, poorly developed markets, harsh climate, and limited access to technological input (e.g. fertilizers, modern farm machinery) face increasing concerns of food insecurity. For example, food production in the African continent has either remained unchanged or decreased between 1960 and 2005 failing to match the needs of a rapidly increasing population. This contrasts with the developed countries where the percapita food production has generally increased even with increase in human population. Millions of resource-poor farmers depend solely on the amount of food produced annually on small pieces of land. The low and meager seasonal crop yields are often insufficient to meet demands for food and other essentials such as purchase of fertilizers, seeds, and farm equipment. Soil erosion impacts on food security in developing countries are confounded by harsh climate (e.g., frequent drought or flooding) and poor socioeconomic and political stability. Increasing demand for food and decreasing crop production in these regions are intrinsically related to the soil's ability to support crop growth and sustain agronomic production.

19.1 Soil Erosion and Yield Losses

Food insecurity affects about 850 million people especially in the tropics and subtropics in sub-Saharan Africa, South Asia, Latin America, Caribbean, and Central Asia (Stocking, 2003). Food shortage is likely to occur in those regions where the largest growth in population is expected (e.g., South Asia, Sub-Saharan Africa). The demand for cereals is expected to increase by 1.3% per year between 2000 and 2025, corresponding to an increase in yield from 2.6 Mg ha^{-1} in 2000 to 3.60 Mg ha^{-1} by 2025 and 4.30 Mg ha^{-1} by 2050 (Lal, 2007). den Biggelaar et al. (2004) synthesized the available information on soil erosion effects on crop yields across the world for soil- and crop-specific conditions and reported that the absolute yield loss ranges between -0.49 and 1.44 $kgha^{-1}$ per Mg of soil lost for grain and legumes, and 0.69 and 127 $kgha^{-1}$ per Mg of soil lost for root crops.

The losses of yield with the same amount of soil loss are smaller in North America and Europe compared to those in other continents (den Biggelaar et al., 2004) (Fig. 19.2). In developing countries, the loss in agronomic production can be much higher than that in North America and Europe per every Mg of soil loss. Accelerated soil erosion is particularly a major problem in Africa where losses of crop production due to erosion range from 2 to 40%, with a mean of 8% for the entire continent (Lal et al., 2000). If soil erosion continues unabated, crop yields are projected to decline by at least 30% in sub-Saharan Africa by 2020 (Fig. 19.3). Since 1970's, rates of soil loss have increased by 10 times faster than the rates of replenishment in the USA and 30 and 40 times faster in China and India. Consequently, about one third of the world's soils has been rendered unproductive (Pimentel, 2000). Although crop yields between eroded and uneroded fields may not always differ in the short term, the costs of production in eroded fields are consistently higher. Crop yields from eroded fields eventually decline with progressive loss of topsoil even with increase in input.

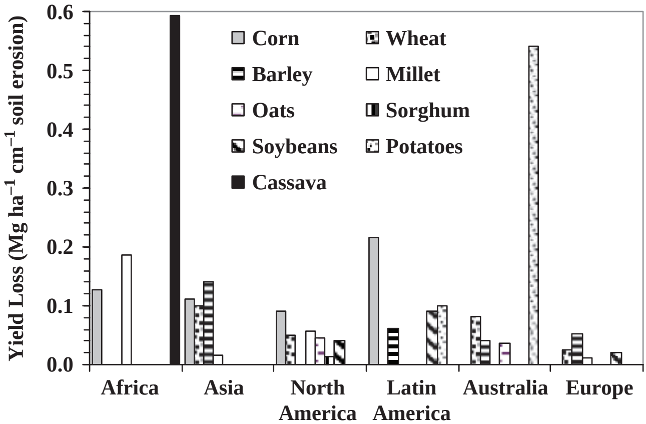
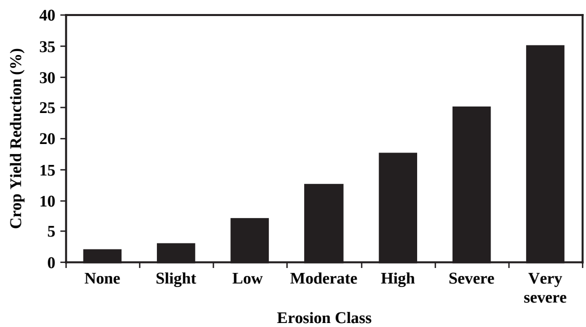


Fig. 19.2 Mean yield loss due to soil erosion for various crops (After den Biggelaar et al., 2004)

Fig. 19.3 Estimated reduction in crop yields in sub-Saharan Africa by 2020 (After Lal et al., 2000)



Soil erosion affects crop production both directly and indirectly (Lal et al., 2000). The direct effects are because of reduction in topsoil thickness, alteration in soil properties, sedimentation and inundation of lowlands, and depletion of soil organic matter and nutrients. The indirect effects of erosion are increase in costs of production because of additional need for fertilizers, pesticides, irrigation, and tillage operations. Tilling exposed hardpans and claypans, repairing ephemeral gullies, and removing sediment from depositional sites increase costs of production. Changing crop rotations and varieties or changing planting and replanting times to offset erosion also require costly input. Accelerated erosion increases the crop's susceptibility to damages by insects and diseases, thereby increasing the use of pesticides.

19.2 Variability of Erosion Impacts

Soil erosion rates vary across soils and ecoregions, as do the erosional effects on crop production. Differences in soil management, cropping systems, conservation measures, and technological input (e.g., fertilizers, organic amendments, lime) determine the magnitude of yield reduction. Unlike the processes and factors of erosion and their effects on soil properties and water quality, the relationship between erosion and crop yield is complex and often masked by the technology (e.g., soil, fertilization). Thus, crop yields vary randomly among years due to fluctuations in climate during the growing season (e.g., air temperature, rainfall amount, intensity, distribution, solar radiation, wind velocity). Crop yields can vary from one season to another even under optimal conditions. This temporal yield variability makes it difficult to precisely characterize and predict erosional impacts on agronomic production. Furthermore, the negative effects of soil erosion on crop yields are often gradual and often go unnoticed until after the soil is severely eroded and is no longer productive.

19.2.1 Soil Type

Similar magnitude of soil erosion can have contrasting impacts on two soils because of differences in their intrinsic characteristics. Crop production on deep soils with high soil organic matter content is affected less by erosion than that on shallow soils with low organic matter content. Crop yields also vary with landscape position (Fig. 19.4). Soils on steep or convex slopes (e.g., shoulder slopes) produce lower yields due to greater losses of soil, thinner topsoil, shallower soil profile, and lower organic matter content, water infiltration rates, and water retention capacity compared to those on footslopes or concave slopes (Cotching et al., 2002). Crop yields are also related to soil order. Because of differences in soil organic matter content and weathering processes, erosion-induced reduction in yield tends to be higher in Ultisols than in Mollisols (den Biggelaar et al., 2001). In Africa, severe

Fig. 19.4 Crop yields are lower in summit and backslope positions than in lower landscape positions due to accelerated erosion (Courtesy USDA-NRCS)

losses of corn yield have been reported on Ultisols, Alfisols, and Vertisols which also have high susceptibility to erosion. Erosion-related reductions in crop yields are minimal on Entisols, which are less susceptible to erosion. In Central and South America, most severe losses of corn yield have been observed on Inceptisols (about 7×10^5 Mg per year) and those of soybean yield on Aridisols. Losses of wheat yield are most severe (or 0.67%) on Inceptisols, Alfisols, and Vertisols in Australia and lowest in Europe (den Biggelaar et al., 2004).

19.2.2 Climate

A similar magnitude of erosion reduces crop yields more in tropical than in temperate climate because of low soil resilience, content of organic matter, and nutrient reserves. Decline in food production affects about 60% of rural population in the tropics and subtropics (Stocking, 2003). Erosion-prone and agriculturally marginal soils are being brought under cultivation in fragile ecosystems under harsh climates (e.g., mountainous areas or drylands) because of the scarcity of prime agricultural lands. Unlike temperate regions, data on the erosion and crop yield relationships are scanty for soils of tropical regions.

19.3 Soil Factors Affecting Crop Yields on Eroded Landscapes

Soil erosion reduces crop production by preferentially removing the nutrient-rich topsoil (A horizons) and by adversely affecting soil structural and hydrological properties. Crop yield, a highly dynamic parameter, is indicative of the spatial and temporal variability in soil properties such as plant available water, nutrient reserves, and soil structure. Three principal causes of crop yield reduction by erosion include: *physical hindrance*, reduction in *plant available water* reserves, and decline in *nutrient supply* (Bakker et al., 2004).

Accelerated erosion reduces crop production by:

- reducing topsoil thickness and rooting depth,
- causing soil compaction and reducing root development,
- inducing surface sealing and crusting which results in reduced seedling emergence,
- reducing the content of soil organic matter and macro- and micro-nutrients,
- exposing subsoil with high clay content and reduced structural stability,
- reducing plant available water capacity,
- decreasing soil macroporosity and aggregation,
- degrading soil chemical properties (e.g., pH, salinity, CEC), and
- reducing water infiltration, hydraulic conductivity, and groundwater recharge.

19.3.1 Physical Hindrance

Accelerated erosion exacerbates problems of surface sealing, crusting, and compaction. Excessively cultivated and eroded soils are truncated and have limited effective rooting depth. Excessive soil erosion exposes subsoil horizons with structural properties unfavorable to crop production. Soils with root restrictive surface and subsurface layers (e.g., claypans, fragipans, hardpans) and those with bedrock at shallow depths are most susceptible to erosion-induced productivity decline. In the southern Piedmont region of the USA, for example, crop yields from eroded soils were reportedly as low as 50% of those from slightly eroded soils primarily because of reduced water infiltration and its storage in the root zone (Radcliffe et al., 1990). The exposed subsoil layers by accelerated erosion are easily compacted, which restricts water and air flux, root growth, nutrient absorption, and thereby reduce yields. In Ohio, corn and soybean yields decreased with increase in rate of soil erosion across four sloping soils (Fahnestock et al., 1995). Corn yield was reduced by about 20% and soybean yield by about 50% although the magnitude of reduction depended on soil drainage, annual rainfall fluctuations, and soil slope (Fig. 19.5).

19.3.2 Topsoil Thickness

The topsoil thickness decreases with increase in soil erosion when its rate exceeds that of soil formation. Thus, mitigating losses by erosion is vital to increasing or sustaining crop yields. All other factors remaining the same, crop growth and yields decrease linearly with reduction in topsoil thickness (Fig. 19.6) (Lal et al., 2000). The topsoil, primarily comprising the Ap horizon, is the physical medium where

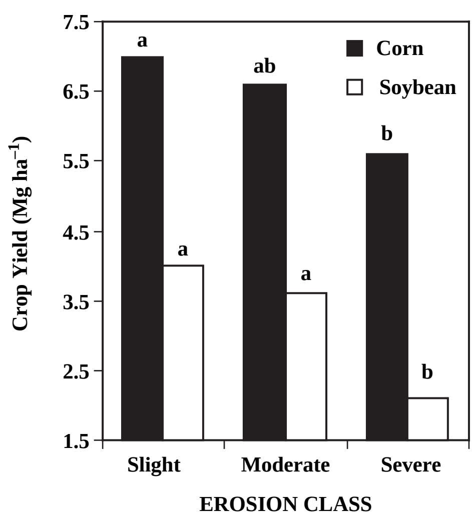


Fig. 19.5 Response of corn and soybean yield to soil erosion in a sloping silty clay loam in Ohio (After Fahnestock et al., 1995). Bars followed by the same letter within the same crop are not significantly different

Fig. 19.6 Water erosion removes topsoil thickness and reduces crop establishment and yields (Courtesy USDA-NRCS)

the largest amount of available water and nutrients is stored and plant roots concentrated. Thus, a complete or partial removal of topsoil adversely affects plant growth. Reduction in rooting depth also increases the sensitivity of plants to anomalies in water, temperature, and nutrient regimes. On artificially desurfaced soil in Nigeria, corn yield decreased by 17 to 65% when 15 cm of topsoil was removed and 38 to 95% when 25 cm was removed (Salako et al., 2007).

19.3.3 Soil Compaction

Eroded soils, with exposed sub-soil horizons, are prone to compaction because of high clay or gravel content, and low organic matter content. Thus, the exposed subsoil generally has higher bulk density and cone index, lower water infiltration rates, and higher runoff losses than uneroded soils. Increase in susceptibility to compaction with acceleration of soil erosion is common in clayey soils with Bt horizon (e.g., Alfisols). Excessive compaction alters soil tilth, limits root growth, and reduces crop production.

19.3.4 Plant Available Water Capacity

Plant available water capacity is one of the principal determinants of crop yield. In general, crop yields decrease with decrease in plant available water capacity of the root zone. Soil erosion reduces plant available water by reducing the topsoil depth and depleting soil organic matter pool. Soils rich in organic matter have high plant available water capacity because of relatively large retention pores. Truncation and exposure of sub-soil decrease available water content because of high cohesiveness and affinity of clay for water. Increase in clay content in eroded soils also decreases nutrient uptake. Shallow-rooted crops are more likely to suffer from topsoil loss than deep rooted crops. Erosion also reduces available water content because of high losses of water by surface runoff.

19.3.5 Soil Organic Matter and Nutrient Reserves

The nutrient-rich fraction of soil organic matter (e.g., particulate soil organic matter) is concentrated in vicinity of the soil surface and is preferentially removed by water and wind erosion because of its low density (Fig. 19.6). The preferential removal reduces the availability of essential nutrients in the topsoil. Higher levels of soil organic matter are associated with higher crop yields because it is a storehouse of macro- and micro-nutrients. Furthermore, soil organic matter regulates pH, CEC, and other processes and properties. Higher soil organic matter enhances microbial processes responsible for nutrient mineralization, solubilization, and recycling. Erosion-induced depletion of soil organic matter reduces structural stability, plant available water, biological diversity, and nutrient supply. Soil organic matter is also essential to improving the soil's ability to retain, store, and recycle water. Excessive soil erosion results in less addition of crop biomass (root/shoot) by progressively reducing the biomass production.

Input of fertilizers reduces erosion-induced nutrient deficiencies. On a tropical sloping soil in Colombia, yields of sorghum, peanut, and cassava decreased abruptly with increase in soil erosion, but yield losses in fertilized plots were much lower than those from unfertilized plots (Fig. 19.7; Flörchinger et al., 2000). Use of chemical fertilizers is not, however, sufficient to regenerate the soil (e.g., biological

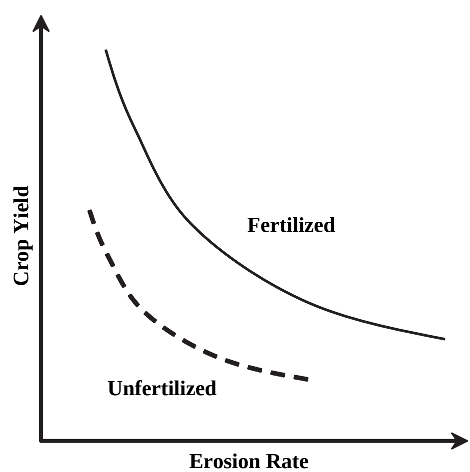


Fig. 19.7 Fertilization impacts on crop yields under eroded soils

properties) and increase crop yields (Jenny, 1980). Crop production is not only influenced by nutrient availability but by a wide range of interactive soil processes (Flörchinger et al., 2000). While fertilizers and organic amendments minimize adverse effects of erosion, the lost soil and its constituents are irreplaceable (Massee, 1990) because decline in soil productivity can not be compensated by chemicals. It takes hundreds of years to regenerate a few millimeters of soil following erosion cessation (Jenny, 1980).

There are two concerns with regards to input of chemical fertilizers. One, the required rate of fertilizers input increases with increase in erosion rates, adding to the production costs (Salako et al., 2007). Uneroded soils respond to fertilization better than eroded soils because of favorable soil structural conditions and higher plant available water. Two, most resource-poor farmers are unable to afford costly fertilizers in economically deprived regions of the world. Promotion and use of locally available amendments (e.g., green manures) are alternatives to inorganic fertilizers (Salako et al., 2007). Organic amendments promote natural soil resilience and enhance microbial activity. There are, however, numerous competing uses of crop residues, animal dung, and other biosolids.

19.4 Wind Erosion and Crop Production

In arid and semiarid regions, wind erosion is a major threat to crop production especially in sandy, loamy-sand, and sandy-loam soils. The magnitude of reduction in crop yield by wind erosion can be as much as that by water erosion or even more. Wind erosion blows away soil organic matter and fine soil particles, adversely affecting soil fertility, structure, and biological properties. Preferential removal of finest and lightest particles results in coarse-textured soils, thereby modifying the textural attributes and the related processes. Sand particles move along the soil surface and are deposited nearby the source. Increase in sand content reduces aggregation, water retention capacity, nutrient cycling and storage, and soil organic matter pool, thereby reducing soil productivity.

Similar to the water erosion, wind erosion also reduces the topsoil thickness and exposes subsoil horizons with textural and structural properties different from those of the uneroded soils. The soil organic matter and fine particles, main reservoir of water and nutrients, are preferentially blown away. Floating dust particles in air consist mostly of clay and humus fractions containing essential nutrients. Windblown clay particles have several times as much nutrients and organic matter as the soil left behind. Strong winds damage crops, reduce seedling growth, increase drought, and reduce crop yields. The abrading sand particles damage leaves and stunt plant growth. Wind erosion also reduces crop production by damaging standing crops through “sand blasting, which reduces vigor and causes de-hydration of young seedlings. Wind erosion is particularly a major constraint in large and unprotected fields devoid of effective vegetal cover.

19.5 Response Functions of Crop Yield to Erosion

Crop production is negatively correlated with the rate of soil erosion, especially in predominantly extractive farming systems. The yield vs. erosion relationships do not always, however, follow a straight line. Crop yields may decrease in a linear, quadratic (concave), logarithmic, exponential, and power or convex function with incremental increase in erosion (Fig. 19.8A through 19.8D). A linear function indicates that crop yield decreases incrementally with an increase in erosion (Fig. 19.8A). Convex response curve portrays a situation where increasing topsoil loss leads to increasing crop yield losses, but the effects of removal of the upper few centimeters of soil is small (Fig. 19.8B). A quadratic or concave function (Fig. 19.8C), in turn, indicates that the removal of the uppermost soil layers has the greatest effect, sharply decreasing crop yields while removal of the lower soil layers cause only minor or no significant reductions in yield.

Knowledge of the shape of the response curve is important to effectively manage eroded soils. Crop yields decrease in a straight line (Fig. 19.8A) or convex relationship (Fig. 19.8B) with an increase in erosion rates suggest that yields increase with further increase in erosion. Erosion in these soils must thus be controlled to minimize risks of severe yield reductions. On the contrary, even a costly restoration of eroded soils with concave yield vs. erosion curves (Fig. 19.8C) may not be feasible because additional increases in soil erosion may have only minor effects on crop reduction. Generally, deficiency in available water and deterioration in soil structural properties produce convex yield-erosion curves, whereas deficiencies in nutrient supply result in linear or concave functions (Bakker et al., 2004). In some soils, topsoil removal may allow plants roots to reach the groundwater zone and offset some of the yield losses due to erosion. In buried soils, removal of infertile topsoil may also improve crop yields (Fig. 19.8D).

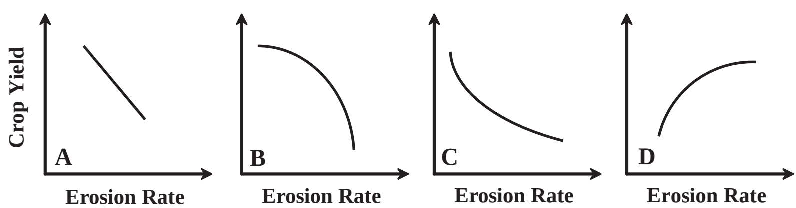


Fig. 19.8 Types of response curves of yield to erosion

19.6 Techniques of Evaluation of Crop Response to Erosion

Monitoring response of crop yields to erosion is complex and a challenging task. Improved agricultural technologies (e.g., fertilization, deep plowing) confound the adverse effects of erosion. Indirect methods are used to assess crop response to *past*, *present*, and *future* rates of erosion (Fig. 19.9). Artificial removal and addition of

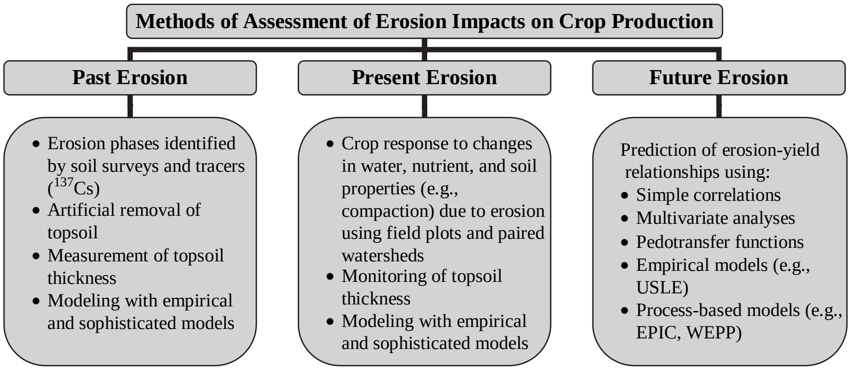


Fig. 19.9 Approaches of assessing erosion impacts on crop yields

topsoil and classification of erosional phases on eroded landscapes are common approaches to assess erosional effects on crop yields. Soil surveys and remote sensing are used to identify lands with exposed subsoil and shallow topsoil depth. Impacts of erosion on crop yields can also be determined by making appropriate measurements on field plots under variable erosion rates. Measurement made on microplots may not produce representative data extrapolable to field or watershed scales.

19.6.1 Removal of Topsoil

Removal of the surface soil at incremental depths is a useful approach to simulate effects of erosion on crop yields (Larney et al., 2000). This method involves use of small and replicated plots under uniform landscape positions. Following scalping, crop yields are monitored under typical tillage operations and managerial inputs. The experimental design consists in mechanically scalping incremental depths corresponding to treatments simulating various degrees of soil erosion. The topsoil is normally removed at 5 cm intervals down to 20 or 40 cm soil depth.

This one-time mechanical truncation of topsoil from small plots does not entirely reflect erosional processes under natural conditions. Soil erosion is a selective process as it systematically sorts out soil primary and secondary particles. It preferentially removes fine inorganic and organic particles and leaves coarse particles and gravels. Furthermore, natural erosion is gradual and does not cause an abrupt disappearance of the whole topsoil unlike the artificial removal. Runoff often concentrates and develops small rills leaving a field with a dissected network of channels, while it inundates the scalped plots. Some of the topsoil remains even in severely eroded fields. The desurfacing method often overestimates the yield losses due to erosion compared to other methods such as field paired comparisons. Thus, data from the desurfacing method must be contrasted with those from field scale studies, and interpreted with a great caution.

19.6.2 Addition of Topsoil

In contrast to scalping, addition of topsoil at incremental depths to eroded soils is complementary method of evaluating the benefits of deep topsoil to crop production. The depth of addition can be 5, 10, 20 or 30 cm, depending on the degree of deposition or the cost of reclamation. Even an addition of 5 cm of topsoil increases crop yields in shallow soils (Larney et al., 2000). Topsoil addition accompanied by proper fertilization or addition of organic amendments is a useful strategy to regenerate severely eroded soils. The added topsoil contains beneficial microorganisms, organic matter, and nutrients similar to the topsoil removed by erosion. Thicker topsoil stores more water and nutrients than shallow topsoil. Adding top soil to exposed subsoil simulates soil development although the reconstructed soil differs from the natural profile of an uneroded soil.

19.6.3 Natural Soil Erosion

Soil erosion vs. crop yield relationships can be established under conditions of natural erosion by using a paired comparison among eroded phases. This is a profound recommended approach and is suited for sloping cultivated fields with marked past erosion imprints. The paired comparison method consists of:

- selecting sloping fields with visible eroded phases under the same soil series, tillage and cropping systems. The soil erosion phases (severe, moderate, slight) are classified on the basis of the thickness of the A and B horizons. Soil aerial surveys coupled with precision agriculture and GIS are used to identify the erosion phases,
- laying out replicated plots across selected landscape positions under varying degrees of soil erosion on the same soil series. Backslope positions are most affected by erosion of all landscape positions. The test plots are often established on backslope positions confining two or more erosion phases,
- applying typical tillage, cropping, and management systems to each paired plot or eroded phase, and
- measuring changes in crop yields and soil properties among erosional phases over time. Plant growth parameters (e.g., seed germination, root distribution, grain and biomass yields) and soil parameters (e.g., nutrient content, plant available water) are measured within each eroded phase.

19.7 Modeling Erosion-Yield Relationships

There are empirical and process-based models to estimate erosion-induced changes in crop yields. There are also mathematical theories of erosion impacts on crop production (Todorovic and Gani, 1987). Empirical models are not as accurate as

Table 19.1 Some site-specific pedotransfer functions for predicting corn grain yield (Y) from soil properties

Predictive equations	r^2
[1]$Y = 0.748 - 1.792SOC + 1.16SOC^2$	0.96*
[1]$Y = 314.5BD^{-24.48}$	0.97*
[1]$Y = 52.51 - 78.64CI + 29.47CI$	0.94*
[2]$Y = 5.5 + 0.92S + 0.211TSD - 0.066(S \times TSD)$	0.70*
[2]$Y = 6.9 + 0.10WSA - 0.84MWD$	0.57*
[2]$Y = 13.3 - 0.38Clay + 0.06Silt$	0.75**
[2]$Y = -12.0 + 0.43Silt$	0.69*
[2]$Y = 17.5 - 0.43Clay$	0.75**
[2]$Y = 3.01 + 56.1N$	0.60**
[2]$Y = 21.8 - 0.133K$	0.58**

[1]Oyedele and Aina (2006) and [2]Lal et al. (2000). SOC (soil organic C content, %); CI (cone index, kg cm^{-2}); BD (bulk density, Mg m^{-3}); S (slope, %); TSD (topsoil depth, cm); WSA (water-stable aggregates, g kg^{-1}); MWD (mean weight diameter, mm); and silt, clay, N (nitrogen), and K (potassium) in g kg^{-1}. *,** significant at the 0.01 and 0.05 probability levels, respectively.

physically-based models. Yet, these are simple and accessible particularly in developing countries where large database required for sophisticated models is not always available.

Simple correlations, pedotransfer functions, and multivariate analyses are commonly used statistical tools to assess erosion vs. yield relationships (Table 19.1). Principal component analysis (PCA) is another approach to identify the most sensitive soil parameters affecting crop yields. Technologies associated with precision agriculture including geostatistics and remote sensing are useful to evaluate soil erosion impacts across large fields. Remote sensing, for example, facilitates the determination of erosional phases and spatial variability of crop yields in complex and eroded terrains. Landsat images are combined with semivariograms and kriging techniques to estimate spatial variability of vegetative cover and crop yields as affected by erosion.

19.8 Productivity Index (PI)

The productivity index (PI) is an empirical approach designed to predict crop yield based on soil properties as affected by erosion including rooting depth, topsoil thickness, organic matter content, and water and nutrient storage capacities. Soils with low rates of erosion, deep profile, high organic matter and nutrient contents, and medium texture have a high PI. The first PI model, developed by Neill (1979), comprised five parameters and assumed that management, climate, and cropping systems remain constant (Table 19.2). Neill's model was slightly modified by Pierce et al. (1983) and Mulengera and Payton (1999). The values of PI range between 0 and 1 with 0 indicating complete restriction of root growth and 1 for maximum root growth.

Table 19.2 Productivity index models. (A = sufficiency of available water capacity; B = sufficiency of aeration; C = sufficiency of bulk density; D = sufficiency of pH; E = sufficiency of electrical conductivity; WF = root weighting factor; r = number of 10-cm increments in the rooting)

PI Models	Parameters
[1] $PI = \sum_{i=1}^{n}(A_i \times B_i \times C_i \times D_i \times E_i \times WF_i)$	Available water capacity, aeration, bulk density, pH, and electrical conductivity
[2] $PI = \sum_{i=1}^{n}(A_i \times C_i \times D_i \times WF_i)$	Available water capacity, bulk density, and pH
[3] $PI = \left[1 - k_y\left(1 - \frac{ET_a}{ET_m}\right)\right]\sum_{i=1}^{n}(C_i \times D_i \times E_i \times WF_i)$	Bulk density, pH, and electrical conductivity, actual crop evapotranspiration, and potential evapotranspiration
[4] $PI = \sum_{i=1}^{n}(A_i \times B_i \times C_i \times WF_i)$	Dry climate $\left(\frac{P}{ETP} < 0.50\right)$: A_1 = available water storage capacity; C = pH Humid climate: $\left(\frac{P}{ETP} > 2.0\right)$: A_2 = aeration capacity; C = soil organic matter Sub-humid to dry climate $\left(0.50 \le \frac{P}{ETP} \le 2.0\right)$: A = the lower value between A_1 and A_2; C = the lower value between C_1 and C_2 B = soil compaction if volume of coarse fragments <30% B = coarse fragments if volume of coarse fragments > 30%

[1]Neill (1979), [2]Pierce et al. (1983), [3]Mulengera and Payton (1999), and [4]Lobo et al. (2005).

The PI models predict about 60 to 80% of variation in crop yield, depending on the site-specific conditions of soil, management, and climate. Use of these models requires extensive validation and adaptation for specific soil-crop-climate conditions. On an erosion-prone soil in Tanzania, the Neill's PI model explained only 47% of the variability in sorghum yield variability while the Mulengera and Payton's PI model incorporating evapotranspiration explained about 87% of the variability (Mulengera and Payton, 1999). The PI models tend to predict crop yields better for dry than wet years (Yang et al., 2003). Thus, soil water dynamics must be incorporated into the PI models to improve their predictive ability.

19.9 Process-Based Models

19.9.1 EPIC

The EPIC is one of the first daily time-step comprehensive models designed to specifically model the impacts of accelerated erosion on crop productivity (Williams et al., 1984). This model is widely used for assessing long-term yield- erosion relationships. It models the impacts of long-term tillage and management on runoff and soil loss and water, nutrient, and pesticide fluxes and their interactive effects on water quality and crop yields. The EPIC simulates erosion-yield patterns across large fields with similar soil and management based on a number of complex set of equations built into a single model called "crop growth model". It integrates information on erosion-affected soil properties with management and climate. Soil depth, strength parameters, water balance factors, temperature, aeration, nutrient availability, and texture are some of the input parameters. Soil profile information for each layer is the data input. The critical plant growth stress factors are considered as follows (Williams et al., 1984):

The water stress factor (WS) is estimated using

$$WS_{=}\frac{\sum_{l=1}^{M} u_{i,l}}{E_{pi}} \tag{19.1}$$

where u is water use in layer l, and E_p is potential plant water evaporation rate on day i. Stress due to poor aeration when soil is near saturation in the upper 1 m of the profile is computed using

$$SAT = \frac{SW1}{P01} - CAF_j \tag{19.2}$$

$$AS_i = 1 - \frac{SAT}{SAT - \exp(1 - 291 - 56.1SAT)} \tag{19.3}$$

where *SAT* is saturation factor, *SW1* is water content (mm), $P01$ is soil porosity (mm), *CAF* is critical aeration factor for crop j, and *AS* is aeration factor.

The *AS* can also be computed as

$$AS = \exp\left[23\,(0.85 - SWF)\right] \qquad SWF > 0.85 \tag{19.4}$$

$$AS = 1.0 \qquad SWF \le 0.85 \tag{19.5}$$

where *AS* is aeration stress and *SWF* is the soil-water factor computed as

$$SWF = \frac{(SW + SW_{15})}{POR} \tag{19.6}$$

where SW_{15} is soil water content at 1.5 MPa, and *POR* is soil porosity of the layer.

Stress due to soil strength (*SS*) is simulated as

$$SS = 0.1 + \frac{0.9BD_l}{BD_l + \exp\left[bt_1 + (bt_2)\,(BD_l)\right]} \tag{19.7}$$

where BD_l is soil bulk density (Mg m^{-3}), and bt_1 and bt_2 are soil texture- dependent parameters. The BD_l is estimated from sand content using pedotransfer functions.

Rainfall/runoff is simulated using three equations: the USLE, MUSLE, and Onstad-Foster modified USLE, which combines both USLE and MUSLE. Unlike other erosion-crop models, EPIC also incorporates information on peak runoff rate, percolation, subsurface flow (e.g., lateral flow, interflow) for a comprehensive simulation of soil hydrological parameters. EPIC also simulates the economic implications of erosion on crop yields and water quality, and it has been linked as submodels to erosion models such as the WEPP and WEPS to simulate plant growth in croplands, pasturelands, and rangelands.

19.9.2 Cropsyst

The Cropping Systems Simulation Model (CropSyst) is another daily time step approach that can simulate crop production across years as influenced by soil erosion (Stöckle and Nelson, 2007). Input data requirements for Cropsyst are similar to those for EPIC. It differs slightly from EPIC in that Cropsyst was developed with a specific focus on crop production, whereas EPIC's main goal was to predict erosion impacts on productivity. Cropsyst has five input files: *simulation control*, *location*, *management*, *crop*, and *soil*. The *Control* file dictates the execution of simulations. It allows the selection of simulations (e.g., nutrients, erosion). The *Location* file stores information on climate (rainfall, air temperature, snow cover, evapotranspiration) and the *Management* file includes information on tillage operations, residue management, and fertilization and irrigation practices. The key input file is the *Crop* file which contains detailed information on crop growth and yield parameters while the *Soil* file, similar to that in EPIC, contains information on runoff characteristics, RUSLE parameters, and soil properties (Table 19.3). The Cropsyst simulates soil water and nutrient budget, above- and below-ground biomass production and decomposition, crop yield as well as soil erosion and fate of chemicals. The model also allows the simplification of simulations reducing model parameterization.

19.9.3 GIS-Based Modeling Approaches

Most of the current crop and erosion models are site-specific and static in nature with regards to management and technological input (Priya and Shibasaki, 2001). Thus, these models fall short of capturing the spatial and temporal dynamics of yield-erosion variability. For example, crop varieties and rotations, fertilizer rates and application dates, irrigation scheduling, harvesting and planting dates remain fixed during simulation using traditional models when, in fact, such variables change over space and time as per climate, management, and other prevailing conditions.

Thus, traditional crop models are now being linked with GIS tools to: (1) account for the dynamics of agricultural management, and (2) expand the applicability of

Table 19.3 Some of the input parameters required for EPIC and Cropsyst models

EPIC		CROPSYST	
Soil and Hydrology	Crop Model	Soil File	Crop File
• Number of soil layers • Soil profile depth • Runoff curve number • Field slope • Slope length • Volumetric water content • Peak runoff rate • Soil porosity • Saturated hydraulic conductivity • Particle size distribution • Organic matter content • Soil strength parameters	• Phenology (daily heat requirement) • Leaf area index • Maximum crop height • Harvest index • Root depth and weight • Aboveground biomass • N and P requirements and accumulation • Drought sensitivity • Crop stage • Water stress factor • Winter dormancy	• Runoff curve number • Number of soil layers • Bulk density • Particle size distribution • RUSLE parameters • Volumetric water content • Saturated hydraulic conductivity • Air entry potential • Cation exchange capacity • pH	• Plant morphology (root depth, leaf area index, root depth, specific leaf area, leaf area duration, canopy cover) • Phenology • Harvest index • Residue decomposition • N requirement and uptake • Salinity level and crop tolerance • CO_2 concentration • Winter dormancy

crop models to larger spatial scales. The GIS-based models have the ability to simulate the spatial variability of yield-erosion relationships across large geographic scales by integrating site-specific information on soil and topographic characteristics, cropping systems, runoff and soil erosion, management practices, and climate. This large scale assessment combines micro- with macro-scale simulations capturing a wide range of conditions of soil, climate, and management systems across regional, national, and continental scales. The GIS collects, stores, and analyzes large sets of data required as input by the erosion-crop models. The GIS also builds maps of model variables such as fertilizer applications, soil texture, pH, irrigation, slope, and others across regions.

The Spatial-EPIC and GEPIC are some of the GIS-based approaches resulting from the combination of GIS with EPIC. The Spatial-EPIC is combined with a GIS environment of 50 km grid size at a country level and 10 km grid size at a regional and global level to estimate the spatial distribution of crop production (Priya and Shibasaki, 2001; Liu et al., 2007). Each grid cell within the GIS-based models simulates crop yield based on site-specific information, and then the modeled results are reverted back to GIS to create maps at various scales of resolution (Fig. 19.10). The GIS software (e.g., ArcGIS) containing the database is combined with data editors to generate the input files for the execution of the EPIC program. The GIS-based models are also being combined with data engines to expand the utility of integrated crop models. These models permit the simulation of future impacts of soil erosion on crop yields and provide a decision making tool for assessing of crop/climate/erosion relationships under variable conditions on a local, regional, national, and global basis.

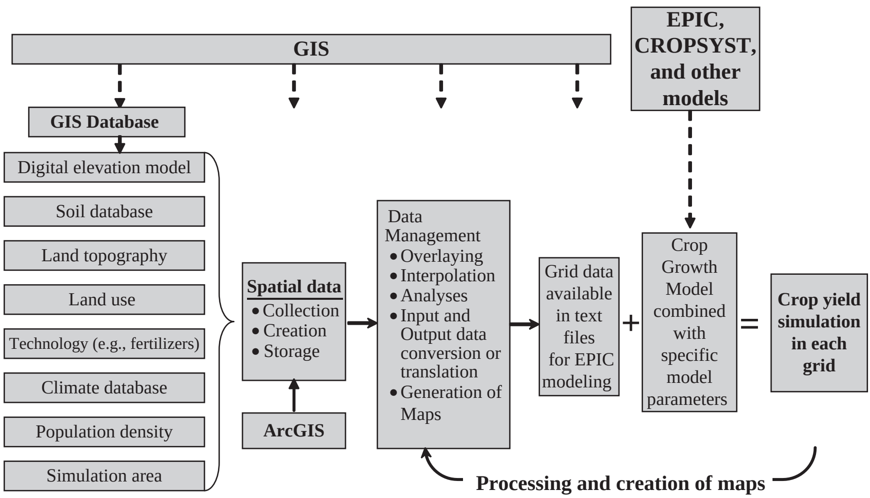


Fig. 19.10 Combination of GIS and process-based models [After Tan and Shibasaki (2003) and Liu et al. (2007)]

Summary

Accelerated soil erosion reduces crop yields and increases concerns over food insecurity. Soil is an indispensable resource to crop production. Any decrease in soil productivity directly and adversely affects food security. Soil erosion impacts on crop production are, however, site-specific. The same amount of erosion can have contrasting impacts in two soils due to differences in depth of topsoil, technological input, soil management, organic matter and nutrient content, slope, and climate. Food production and availability are not the same in all regions. While crop yields have increased in industrialized countries attributed to rapid agricultural mechanization and technological input in recent decades, yields have remained stagnated or decreased in developing countries. Presence of degraded soils and limited access to technology (e.g. fertilizers, herbicides, modern farm machinery) are the main factors for the reduced crop yields in developing regions. This low crop production threatens the food security for the increasing population in poor regions such as in sub-Saharan Africa, Latin America, Caribbean, and Central Asia where food production is three to six times lower than that in developed countries per every Mg of soil loss.

Soil erosion reduces crop yields by decreasing topsoil thickness, degrading soil properties, and inducing losses of soil organic matter and nutrient contents. Soil loss and nutrient mining by crops are the direct causes of the decrease in crop yields.

Erosion reduces rooting depth, induces surface sealing and crusting, causes soil compaction, and reduces seedling emergence and thus crop yields. Crops yields decrease with increase in soil erosion in a linear, quadratic, logarithmic, exponential, and power/convex function. There are various techniques to assess the erosion impacts on crop yields. Removal and addition of topsoil and monitoring crop yields across erosion phases using soil surveys and remote sensing are some approaches to assess erosion effects. Empirical and process-based models (e.g., EPIC, CROPSYST), statistical tools (e.g., multivariate analysis) are used to estimate erosion-induced changes in crop yields. The productivity index is a widely used approach to estimate crop yields from soil properties (e.g., rooting depth, topsoil thickness, organic matter content, water and nutrient storage capacity).

Study Questions

1. What are the mechanisms by which soil erosion affects crop production?
2. Describe the process-based models used for estimating crop yields as affected by erosion.
3. Describe the methods used for measuring erosion impacts on crop yields.
4. Explain the processes that explain differences in crop yield vs. erosion curves.
5. Describe the variability of crop yields with the landscape positions.
6. Discuss how changes in soil physical properties due to erosion affect crop yields.
7. Show an example of the use of productivity index (PI).
8. Discuss the direct and indirect effects of erosion on crop yields.
9. Explain the reasons for the site-specificity of erosion impacts on crop productivity.
10. Discuss differences between EPIC and CROPSYST models.

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Chapter 20
Climate Change and Soil Erosion Risks

The widely recognized global climate change is attributed mostly to anthropogenic activities which are the leading sources of greenhouse gases (GHGs) (IPCC, 2007). The current concentration of atmospheric GHGs is the highest level since 650,000 yr (IPCC, 2001). Principal GHGs, both natural and synthetic, are listed in Table 20.1. Global warming potentials (GWPs) are often used to compare the abilities of GHGs to trap heat and warm the atmosphere. The GWPs are estimated from the heat-absorbing ability and the decay rate of each GHG with respect to CO_2.

The CO_2 is the largest fraction of GHGs and represents about 60% of the total radiative forces (Rastogi et al., 2002). The rate of CO_2 emissions has accelerated since 1995, and the average rate of annual increase for the decade ending in 2005 was 1.9 ppm per year compared to the annual rate of 1.4 ppm between 1960 and 2005 (IPCC, 2007). Between 1750 and 2005, the atmospheric abundance of *CO_2 increased by 31% (280 ppm vs. 379 ppm), CH_4 by 151% (715 ppb vs. 1774 ppb), and N_2O by 17% (270 ppb vs. 319 ppb)* (Lal, 2006; IPCC, 2007).

Combustion of fossil fuels is a primary source of CO_2. Changes in land use (e.g., deforestation, soil tillage, fertilize use, biomass burning, removal of crops residues) also contribute to emission of GHGs. Agricultural activities are prime sources of CH_4 and N_2O emissions. The current rate of CO_2 emissions from fossil fuel combustion exceeds those from land use change from native vegetation to agriculture,

Table 20.1 Some of the GHGs responsible for global warming (After IPCC, 2001)

Name of GHG	Global warming potentials (GWP)
CO_2	1
CH_4	23
N_2O	296
Hydrofluorocarbons (HFCs)	
• HFC-23	12,000
• HFC-134	1,300
• HFC-152	120
Perfluoromethane (CF4)	5,700
Sulfur Hexafluoride (SF6)	22,200

but the total contribution of land use conversion to GHG emissions since the dawn of settled agriculture is high (Ruddiman, 2003).

20.1 Greenhouse Effect on Climatic Patterns

The progressive build-up of GHGs in the atmosphere and the attendant climate change have altered the energy balance of the earth (SWCS, 2003). These changes have altered precipitation and temperature patterns globally, and changes in climate are projected to be more drastic in the near future (IPCC, 2007). The greenhouse effect is the process by which outgoing longwave solar radiation is trapped by the GHGs, causing the earth's warming. Increase in ocean temperatures, melting of ice and snow from glaciers and snow capped mountains, and sea level raise are consequences of global warming (IPCC, 2007). Some regions are becoming drier due to either decreases in precipitation or increases in evapotranspiration rates whereas others are receiving increasingly intense and unevenly distributed rain storms.

Sharper or abrupt shifts in temperature and rainfall patterns are also predicted in the near future (Wigley, 2005), which can have greater implications than gradual changes. Simulation models predict that any future increase in GHGs concentrations over the current levels are expected to cause more drastic effects on climate change. The warmer climate in the 21st century may increase precipitation and runoff rates in humid regions and cause water shortages and drought stress in dry regions (Yuan et al., 2005). Quantity and quality of water resources are directly affected by changes in precipitation and temperature regimes.

20.1.1 Temperature

Average temperature of terrestrial and aquatic ecosystems is progressively increasing. Between 1995 and 2006, 11 out of 12 yr were on the average warmer than previous years. Mean air temperature increased by 0.76 °C between 1850 and 2005 (IPCC, 2007). Increases in temperature during the last 50 yr (1955–2005) were twice as high as those in the past 100 yr (1855–1955). In accord with the change in air temperature, trends of increases in both maximum and minimum soil temperatures are also consistent. Under the present business as usual scenario of GHGs emissions, it is predicted that the temperature will increase by 2 °C by 2100 at a rate of about 0.2 °C per decade, while the sea level would rise at the rate of about 25 mm per decade (IPCC, 2007). Cold days and seasons are gradually being replaced by hot days and seasons in the high latitudes. The build-up of GHGs in the atmosphere has long-term consequences. Even if the GHGs emissions were completely halted, the present levels of GHGs would continue to increase the temperature by about 0.1 °C per decade over the next few decades due to the "thermal inertia" which signifies that the today's level of GHGs concentration has not yet fully displayed its impacts on the climate (IPCC, 2001).

20.1.2 Precipitation

Global warming is changing the patterns of rainfall distribution across the globe. Total precipitation has either increased or decreased with global climate change depending on the latitude (Fig. 20.1). Mean precipitation is increasing in high latitudes and tropical areas but is decreasing in sub-tropical and arid regions. It is estimated that rainfall fluctuations will be more dramatic during the 21st century. Mean annual precipitation has already increased by about 6% between 1910 and 1999 in the USA (SWCS, 2003). Other estimates show increasing trends of annual precipitation between 1900 and 1998 by 10 to 40% (Groisman et al., 2001). Snow cover in springtime in western USA has decreased while intense precipitation events have increased in the eastern USA (Groisman et al., 2001).

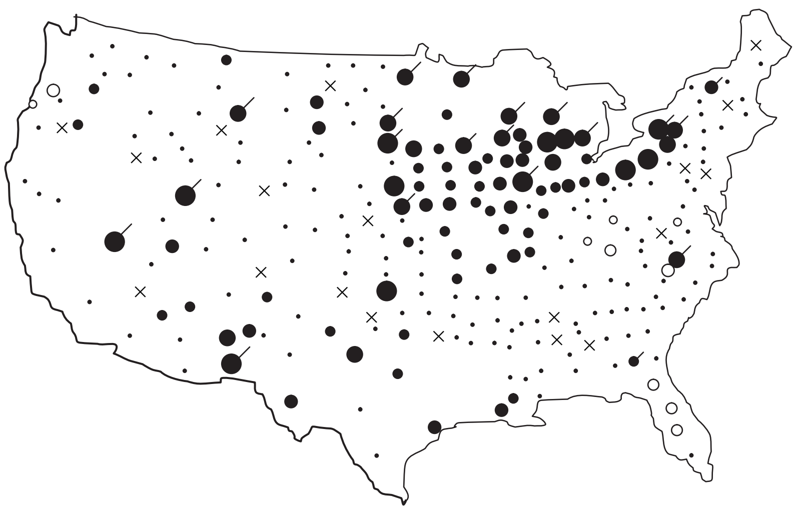

Fig. 20.1 Frequency of intense precipitation events in the USA between 1931 and 1996 (After Kunkel et al., 1999). *Black dots* signify upwards trends in percentage of precipitation whereas *white dots* signify downward trends. *A tail attached to a dot* indicates a significant trend at the 0.05 probability level. The “x” is for regions without complete precipitation records

20.1.3 Droughts

Precipitation amount has decreased in arid and semiarid regions of the world such as in parts of Africa, Asia, and parts of South America (IPCC, 2007). This decrease is responsible for the frequent droughts in recent decades. Drought periods and intensities have increased since 1970's due to higher temperatures, lower precipitation amounts, higher evaporation rates, and stronger wind storms. Risks of occurrence of drought and flood are likely to increase depending on the latitude. In drylands, soil productivity is expected to decrease because of both drought and heat stress.

20.1.4 Other Indicators of Climate Change

Flood frequencies, heat waves, wind storms, tropical cyclones, and other extreme weather events have increased in recent years. The water vapor content and salinity levels of the oceans have increased, and the sea level has risen over the last few decades. While the average sea level increased by 1.8±0.5 mm per yr between 1861 and 2003, it increased by 3.1±0.7 mm per yr between 1993 and 2003 (IPCC, 2007). The ocean's capacity to absorb C may decrease, which will be another major contributor to global warming. The decrease in ocean C sink could increase global temperature by 1 °C in 2100 (Reay, 2007). Arctic sea ice is shrinking by about 3% per decade since 1978 due to increase in air temperature (IPCC, 2007). Glacial lakes have expanded in size and number in recent decades due to sustained melting of snow and ice cover and soil thawing of permafrost in arctic and boreal regions.

20.2 Climate Change and Soil Erosion

The predicted climate change is expected to increase risks of soil erosion, which can exacerbate soil degradation and desertification (Lal, 2006). The magnitude of this expected increase in water and wind erosion risks will most likely depend on local and regional conditions (O'Neal et al., 2005) (Table 20.2). It may increase by soil erosion 5 to 95% and runoff loss by 5 to 100% in agricultural lands (SWCS, 2003). Changes in precipitation patterns, in interaction with those of land-use, vegetative cover, and soil erodibility affect the erosion rates (Fig. 20.2).

20.2.1 Water Erosion

According to the available measured and simulated data, runoff and soil erosion are likely to increase with increase in precipitation under the new climate. The intensification of rain storms could increase water erosion because changes in intensity have a greater influence on soil erosion than frequency and amount of rainfall (Table 20.2). The proportional increase in water erosion may be even more than the relative increases in precipitation in some regions due to positive feedbacks (e.g., increase in soil erodibility) (Fig. 20.2).

Average annual runoff rates are expected to increase by 30 to 40% in high latitudes, and decrease by 10 to 30% in arid and semiarid regions prone to drought

Table 20.2 Effect of the predicted increases in annual rainfall amount on soil erosion in the 21st century

Location	Scenario	Model	Rainfall increase (%)	Soil erosion increase (%)
UK South Downs[1]	High levels of CO_2 emissions	EPIC	5	10
			10	26
			15	33
U.S. Corn Belt region[2]	Corn/soybean rotations	EPIC	20	37
Eastern U.S. Corn Belt region[3]	A range of cropping systems, soil types, and climate conditions	WEPP-CO_2, Hadley Centre (HadCM3-GGa1), and Decision Support System for Agrotechnology Transfer (DSSAT)	10–20	241
Eight locations in the USA[4]	A wide range of soils and rainfall intensities	WEPP and RUSLE	1	2.4
USA[5]	Review of modeled results across USA	WEPP and HadCM3-GGa1	1	1.7

[1]Boardman and Favis-Mortlock (1993); [2]Lee et al., (1996); [3]O'Neal et al. (2005); [4]Pruski and Nearing (2002); [5]Nearing et al. (2005).

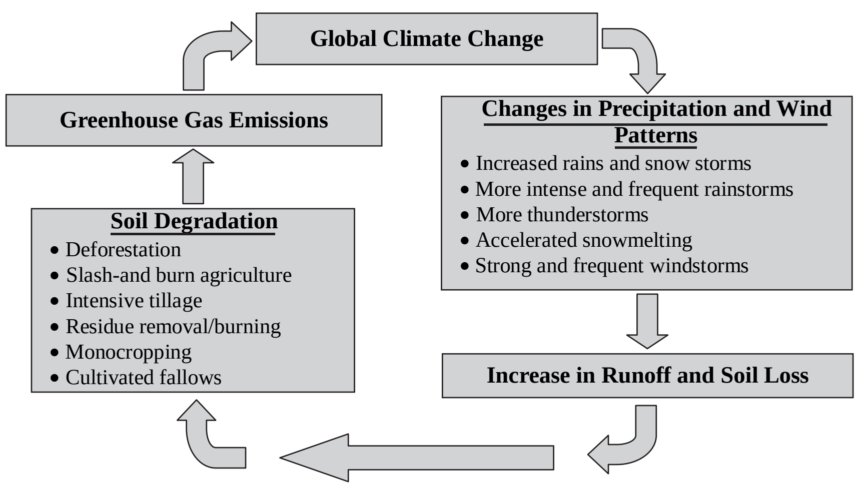

Fig. 20.2 Interactive effects of land use conversion on climate change, runoff and soil erosion

stress (IPCC, 2007) (Table 20.2). Runoff rates are projected to significantly increase in high latitudes and southeast Asia, and decrease in central Asia, Mediterranean, Southern Africa, South America, and Australia by 2050 with a 1% increase in CO_2 concentration (IPCC, 2001). Increases in runoff and soil erosion risks are likely to be high in coastal areas and disturbed agricultural lands. Intense rainstorms and flash floods are expected to contribute to increased runoff losses in northern latitudes. Simulations of the various erosion factors in the Midwest USA predict an increase in erosion rates by about 85% over the next 50 yr.

Site-specific dependence. Effects of climate change on erosion are expected to be more severe in soils managed by resource-poor farmers in developing countries. Such high risks in developing countries may be attributed to large areas of degraded ecosystems and the fact that erosion control strategies are either limited or non-existent. In the humid tropics, heavy rainfall would increase runoff and cause flooding of lowlands. While the amount of rainfall may decrease in arid and semiarid regions, the erosivity of rains may increase. Thus, decrease in precipitation rates may not always result in lower soil erosion rates. Indeed, predicted decreases in rainfall in arid and semiarid regions are just as likely to increase water erosion as in humid regions (Pruski and Nearing, 2002). High losses of runoff water are expected from sloping lands and undulating terrains.

Landscape stability. Sedimentation of downstream water bodies with runoff sediment is a concern under the new climate. Landslides, streambank erosion, and mudflows may increase under saturated and concentrated runoff conditions in sloping lands. Increased runoff and soil erosion can also increase formation of ephemeral and permanent gullies, affecting landscape characteristics and channel/waterway stability. Intense rain storms and high volumes of runoff can develop concentrated flow erosion in farmlands. High runoff volume in interaction with field topography often causes runoff to concentrate in natural swales as runoff moves downslope, which is known as concentrated flow erosion. Contribution to concentrated flow erosion to total erosion can become increasingly significant with increase in rainfall intensity and frequency.

20.2.2 Nutrient Losses in Runoff

The projected increases in soil erosion by climate change can cause pollution of water resources with dissolved and suspended loads. Delivery of dissolved sediment in runoff to downstream water increases with periodic increase in precipitation (SWCS, 2003). Because soil warming stimulates decomposition and mineralization of soil organic matter, more soluble nutrients and soil-borne chemicals may be released in water runoff. Thus, there is a greater chance for the soluble compounds to be delivered to surface and subsurface water bodies through leaching and runoff. On a forested catchment in Norway, runoff from artificially warmed field plots had greater concentration of nitrates (NO_3-N) and ammonia (NH_4-N) than that from control plots without warming (Lükewille and Wright, 1997). Excessive nutrient losses can cause eutrophication and acidification of water sources. Earthworm burrows and other biochannels may also cause preferential or bypass flow of water with chemicals into the lower horizons (Shipitalo and Gibbs. 2000.). Soil cracks are also pathways for bypass flow of rainwater and dissolved chemical. This bypass flow can increase the risks of water surface and subsurface pollution with nutrients and pesticides.

20.2.3 Wind Erosion

Wind erosion is likely to increase linearly with reduction in precipitation and increase in temperature in arid and semiarid regions. The increase in air temperature may increase evaporation rates and reduce soil water content as the soil heats up, thereby reducing vegetative cover and biomass production. These conditions can favor an increase in the velocity and erosive power of the winds, increasing rates of wind erosion (Lee et al., 1996). In dry regions, climate change may, in some cases, reduce water erosion but drastically increase wind erosion.

Increase in duration and intensity of dry seasons accompanied by strong winds can exacerbate the wind erosion risks. Indeed, losses of soil by wind have already increased in drylands because of high wind erosivity. About 25% of cultivated lands in arid and semiarid regions are severely affected by wind erosion. In China, strong sandstorms have increased from 5 times in the 1950's to 20 times per year in the 1990's (Ci, 1998). In northern China, increase in temperature by 1°C increased average wind erosion by 31 Mg km^{-2} yr^{-1}, and that land use change was a main driver of the wind erosion risks (Gao et al., 2002). While it is possible that increase in CO_2 concentrations may partly offset wind erosion attributed to increase in biomass production and vegetative cover (Lee et al., 1996), the lower precipitation rates and higher evaporation rates can cause heat and drought stress and reduce plant production

20.3 Complexity of Climate Change Impacts

Impacts of the projected climate change on soil erosion are expected to be complex and variable, depending on ecological, landscape, management, and climatic characteristics. Precipitation and temperature patterns are as variable as their effects on soil erosion. Some soils are more vulnerable to erosion than others. Current prediction of climate change effects on soil erosion is subject to uncertainty due to the many interactive processes including rainfall erosivity, soil erodibility, and vegetative cover and landscape characteristics. In some regions, even small changes in precipitation may result in large increases in runoff and soil erosion due to the interactive effects, while in others greater vegetation production from higher temperature and rainfall amounts may actually decrease rates of water and wind erosion. A 2°C increase in temperature and 10% decrease in precipitation would reduce runoff by 20%, while a 4°C increase in temperature and 20% decrease in precipitation would reduce runoff by 30% (Nash and Gleick, 1991). On sandy soils in northern China, an increase in temperature by 1°C reduced the average rate of water erosion by 5 Mg km^{-2} yr^{-1} (Gao et al., 2002).

20.4 Erosion and Crop Yields

The significant reduction of topsoil thickness by soil erosion under the new climate can reduce crop yields depending on site-specific conditions (Boardman and Favis-Mortlock, 1993). Erosion removes the well-structured and organic matter-rich soil layers, thereby reducing soil fertility, deteriorating soil structure, reducing water retention capacity, and adversely affecting crop production. In some soils, the rate of soil erosion under the projected climate change may be too high to be compensated by the slow natural rate of soil renewal. The adverse impacts of soil erosion may be small on well-developed soils with thick horizons and high soil organic matter content, and large in shallow and stony soils with low inherent fertility and shallow effective rooting depth. Extreme runoff events can adversely impact soil resilience and cause irreversible damage to soil quality.

Impacts of climate change-induced soil erosion on crop yields may be complex and often detectable only over long time periods. Over a sort-time horizon, negative impacts of erosion on crop yields are often masked by high input of fertilizers and use of improved varieties. The high chemical input agriculture, however, may also accentuate non-point source pollution. Higher soil erosion causes on-site and off-site adverse impacts. Losses of topsoil and crop yields are some of the on-site effects while water pollution and inundation downstream are some of the off-site effects.

20.5 Impacts of Climate Change on Soil Erosion Factors

20.5.1 Precipitation

Rain storms are more intense than ever (Nearing et al., 2005). While the annual precipitation amount has remained relatively constant in some regions, the number of intense storms has increased particularly in North and South America and northern Europe (SWCS, 2003). More intense rainstorms occurred between 1970 and 1999 than between 1910 and 1970. In the USA, total precipitation increased between 1910 and 1996 and that 53% of it corresponded to intense events. Over the 21st century, the frequency of intense events will increase by about 20 to 60% (SWCS, 2003).

Detachment and transport of soil by water depend on the amount and intensity of rainfall. Model results show that rainfall erosivity is projected to increase significantly with the projected climate change. Rainfall intensity is a stronger determinant of erosion than rainfall amount and frequency. Soil erosion is a function of rainfall erosivity, which is computed as the product of total rainstorm energy and maximum 30-minute intensity (SWCS, 2003). Even small amounts of rain can cause large amounts of soil loss if rain is in the form of intense storms. An increase of 1% in total rainfall would increase soil erosion rates by 1.7% if rainfall intensity increases correspondingly, whereas it would increase erosion rates by only 0.85% if intensity remains unchanged (Pruski and Nearing, 2002). Thus, increases in erosive energy of rainfall due to climate change may strongly increase soil erosion rates.

Rainfall exerts both *positive* and *negative effects* on soil's resistance against erosion. It negatively affects soil stability by increasing detachability and transportability of soil particles in runoff, and positively by increasing soil water content and promoting growth of plants and the vegetative cover. The higher biomass production protects the soil from the erosive energy of rainfall and reduces runoff and soil loss.

The positive effect of rainfall on biomass production may be, however, counteracted by the higher temperatures which would accelerate evaporation rates, reduce plant available water content, and increase temperature and drought stress.

Increases in air temperature may lead to faster rates of snow melting, which can result in higher snowmelt runoff and soil erosion. Snowpacks are fragile and highly sensitive to temperature increase. Changes from snowfall to rainfall due to increased temperature may be another immediate source of storm runoff for exacerbating soil loss. Snow storms are likely to be replaced by rain storms (Nearing et al., 2005). It is projected that winter runoff will be higher than summer runoff due to early snowmelting and higher rain to snow ratio.

20.5.2 Soil Erodibility

Changes in water and temperature regimes may significantly impact soil processes and properties with the attendant effect on soil erodibility. Degradation of soil structure reduces macroporosity and water infiltration rates. Soils with degraded structure have the greatest losses of soil by water and wind erosion. Changes in near-surface soil conditions (e.g., crusting, surface sealing, and compaction) are climate change-induced processes that can also increase soil erosion. Soil aggregate stability may decrease with increase in temperature and decrease in rainfall due to changes in vegetation cover (Cerda, 2000). Shifts in land use, crop varieties, tillage methods, and plant species in response to change in climate are also likely to affect soil erodibility. Interactive effects of land use and management and soil type determine soil structural stability, soil organic matter content, and water infiltration rates, which are important parameters of soil erodibility (Fig. 20.3). Climate change may not only affect surface runoff but also water movement through the soil.

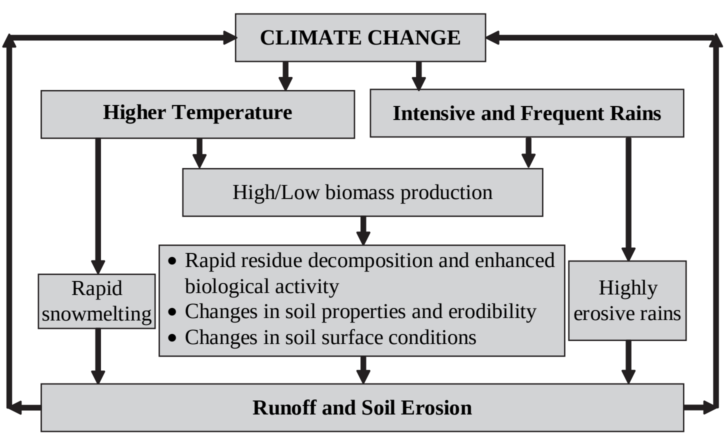

Fig. 20.3 Complex interrelationships between climate change and soil erosion

20.5.3 Vegetative Cover

Changes in vegetation cover as a result of shifts in land use and management may have a large impact on soil erosion under the changing climate (Nearing et al., 2005). Clear-cutting of forestlands, for example, eliminates the vegetative cover and can exacerbate soil erosion under increased rainfall intensities. Both canopy and soil surface residue/litter cover are essential to reducing water and wind erosion. Runoff and soil loss increase with decrease in percent vegetation cover. The probable increase in biomass production under climate change may provide greater vegetative cover and reduce soil erosion in northern latitudes. Increasing temperature and decreasing rainfall in tropical regions may change the vegetation from deciduous tree and rainforests to grasslands or savannas, whereas decreasing temperatures and increasing rainfall in subhumid regions may favor production of trees (Nearing et al., 2005). Arid regions with sparse vegetation would support less vegetation and become more desertified under higher temperature and lower rainfall, thereby resulting in greater wind erosion.

20.5.4 Cropping Systems

Global climate change may affect the choice of cropping systems in response to changes in precipitation and temperature. Farmers most likely will react to climate change by shifting cropping systems (e.g., crop rotations, varieties) and dates of planting to adapt to new climatic conditions. Early soil thawing and warming due to higher temperature may permit farmers to plant crops at earlier dates than usual in order to avoid high temperatures during silking or to accommodate two crops in one year. This shift in planting dates would increase the length of cropping seasons. The effectiveness of cropping systems for reducing water and wind erosion depends on the percentage of canopy cover and the amount of residue left at harvest. Replacing high-biomass with low-biomass producing crops may increase soil erosion. In the Midwest USA, shifting crop rotations (e.g., from corn and wheat to soybeans) in interaction with higher precipitation is projected to increase soil erosion by about 300% in 2040–2059 with respect to 1990–1999 (O'Neal et al., 2005). Replacement of corn-soybean rotations with continuous soybean may exacerbate soil erosion due to reduced biomass production (Southworth et al., 2002a).

20.6 Soil Formation

Climate change is projected to impact soil development and behavior because climate is one of the key factors of soil formation. Changes in soil properties are a function of independent variables as per Eq. [20.1] (Jenny, 1941)

$$S = f(CL, O, P, R, T, \ldots\ldots\ldots\ldots\ldots) \tag{20.1}$$

where S is soil property, CL is climate, O is soil biota, P is parent material, R is relief or topography, and T is time of soil formation. Climate is the most active and influential factor because it covers large geographic areas. Precipitation, temperature, relative humidity, and solar radiation are the main climatic factors influencing soil weathering, horizonation, and profile development. Precipitation and temperature control processes of soil development such as rates of evaporation, water storage, illuviation, leaching, and biological activity. Soils are more weathered in regions with high precipitation and temperature. The local predominant climate determines the rate at which soils develop and degrade in response to soil management. Thus, rate of soil formation is strongly correlated with changes in climate.

Climate does not act alone but in interaction with other soil forming factors such as vegetation, soil organisms, landscape characteristics, parent material, soil management, and time (Fig. 20.4). For example, climate determines decomposition and accumulation of soil organic matter, storage of water, biomass production, and activity of soil organisms, which, in turn, affect soil development. Climate also moderates rates of nutrient uptake, seed germination, and root development. It controls diversity, number, growth, and activity of soil macro- and micro-organisms.

Desert regions with high temperatures and low precipitation possess sparse vegetation, and the limited biomass produced is rapidly decomposed. In contrast, cold and wet regions accumulate organic matter, and bacterial activity is suppressed during cold seasons. In humid and hot regions (e.g., tropics), the decomposition of organic matter is rapid due to high precipitation and temperature, resulting in reduced accumulation of nutrients and soil organic matter.

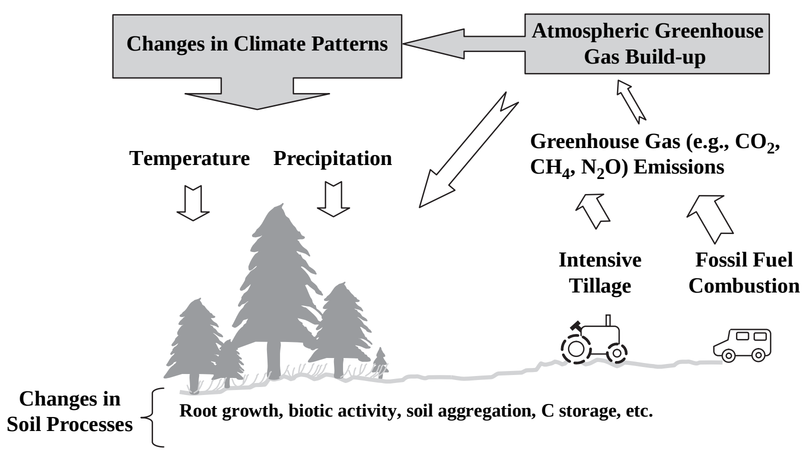

Fig. 20.4 Interrelationships among CO_2 emissions, climate change, plant growth, and soil properties

20.7 Soil Processes

Climate change can influence the type and magnitude of processes of soil formation (Table 20.3). Rate of physical, chemical, and biochemical *weathering* may increase with increase in temperature. *Physical weathering* is more predominant than *chemical weathering* in dry and hot regions. Precipitation determines the translocation of colloidal particles and ions to deeper horizons by *elluviation* and *illuviation*. *Leaching* of dissolved chemicals may increase with increase in rainfall under the new climate. Gradual loss of bases (e.g., Ca^{2+}, Mg^{2+}, Na^{+}) from the topsoil by leaching causes *acidification*. Soils with lower pH may require larger application of lime to sustain crop production. Soil temperature influences chemical processes by altering dynamics of soil solution, diffusion of chemicals, solubility of organic and inorganic materials, and establishment of dynamic and static equilibrium conditions (Buol et al., 1990). Reaction rates, diffusion, and solubility of salts increase with increase in soil temperature. Changes in soil chemical processes from small changes ($<5^\circ$ C) in temperature may be detectable only after several years.

Table 20.3 Impacts of temperature and precipitation change on soil processes

Temperature	Precipitation
• Evapotranspiration rates	• Leaching and drainage
• Organic matter decomposition	• Illuviation and eluviation
• Biological activity	• Runoff
• Nutrient and water uptake by plants	• Lateral flow and interflow
• Wetting and drying, and freezing and thawing	• Nutrient uptake and solubility
	• Acidification
• Reactions of ions in the soil solution	• Aqueous reactions
• Solubility of salts and organic compounds	• Hydrolysis and hydration
• Chemical equilibrium, and reductions and oxidations	

In arid and semiarid regions, evaporation may exceed precipitation. Under excessive evaporation, soluble salts in saline water move upward by capillarity and accumulate in the soil surface, causing *salinization*. Increase in temperature and decrease in precipitation by climate change could promote formation of *saline and sodic soils* in warmer and drier regions of the world. Rise of sea levels with climate change can also expand saline areas along coastal areas.

20.8 Soil Properties

20.8.1 Temperature

Changes in soil temperature are directly correlated with air temperature. Soil surface temperature tends to be slightly higher than the air temperature. In temperate regions, it can be about 1.1° C higher than the air temperature (Buol et al., 1990).

Increases in soil temperature may warm up and thaw perpetually frozen soils, thereby increasing land available for cropping in northern latitudes. Increase in soil temperature may also cause creeping of croplands northwards and expand crop growing seasons.

20.8.2 Water Content

Soil water content is a function of the quantity of precipitation. Thus, even small changes in precipitation by climate change would influence soil water regime through changes in biomass production. Increasing air temperature by climate change is projected to accelerate evapotranspiration and water use by plants, thereby reducing water storage. A study in Ohio showed that soil water content decreased with increase in soil temperature as a result of crop residue removal from no-till fields (Blanco-Canqui and Lal, 2007).

20.8.3 Color

Increasing soil temperature can accelerate soil weathering and decomposition of humus. Global warming may tend to develop redder soils (e.g., Oxisols, Ultisols) although this process occurs under large increases in temperature are long periods. Floods and waterlogged conditions can also affect soil color by affecting drainage rates.

20.8.4 Structural Properties

Soil structure is also sensitive to changes in soil temperature and amount of precipitation (Sarah, 2005). The change in climate may affect soil structure indirectly by altering air temperature, precipitation, plant growth, and microbial activities. Changes in soil structure under the projected climate change can have major implications because soil structure:

- determines the rate at which rainwater infiltrates into the soil or becomes runoff,
- determines the soil's resistance to the erosive forces of water and wind,
- moderates fluxes of water, air, and heat,
- influences decomposition of soil organic materials and activity of soil organisms,
- stores soil organic C and determines the fate of organic materials, and
- absorbs, buffers, and degrades pollutants.

Temperature effect. Increase in soil temperature is most likely to influence soil structure by altering soil physical (e.g., shrink-swell, freeze-thaw) and biological (e.g., organic matter decomposition, microbial activity) processes. Experiments simulating impacts of soil warming show that aggregate stability decreases with increase in soil temperature due to rapid organic matter decomposition and decrease in microbial processes (Rillig et al., 2002). Aggregate stability and strength are adversely affected by fluctuations in wetting/drying and freezing/thawing decrease. Higher temperature may also cause desiccation of soils and formation of cracks, and cause a problem to building structures and crop production. Poorly drained, wet, and clayey soils could, however, benefit from moderate drying. High temperature can cause excessive drying of soils and result in compaction.

Precipitation effect. Intense rainstorms may deteriorate surface soil structure and increase soil erosion. Soil detachment rates increase with increase in rainfall erosive power. Higher soil erosion from higher precipitation rates reduces topsoil thickness and exposes soil horizons with reduced structural development. Increase in rainfall amounts with low intensity can, however, improve soil structural stability by enhancing above- and below-biomass production (Cerda, 2000). In arid and semiarid regions, reduction in rainfall amounts from climate change can deteriorate soil structure by reducing vegetation growth. Soils with high clay content have high shrink and swell potential and can respond rapidly to changes in precipitation. Excessive rain can also make soils too wet for cultural operations. Soils that are too wet are structurally unstable and susceptible to compaction by traffic.

CO_2 concentration effect. The high atmospheric CO_2 concentration is expected to affect soil structure by increasing the above- and below-ground biomass production. It can affect both quantity and quality of the biomass produced. Plant residues with high C:N ratio (low N content) are prone to slow decomposition and thus improve and maintain soil structure (Young et al., 1998). The positive effects of high CO_2 concentration on soil structure through increased biomass production may be, however, offset by the higher temperature that would reduce plant available water plant and cause soil desiccation.

20.8.5 Soil Biota

Projected climate changes may affect soil, plant, and animal interactions. Soil organisms rapidly react to abrupt changes in temperature. Changes in biomass production under the new climate may affect the number, activity, and diversity of soil organisms (Lavelle et al., 1997). Global warming can favor the proliferation of earthworms, termites and other organisms. These organisms play a major role in decomposing organic matter as they devour coarse organic matter and determine its distribution and rates of decay. By decomposing organic matter and releasing organic binding compounds, earthworms, termites, and related soil organisms influence soil structural properties and soil erodibility. Soil structure shelters and provides an environment to organisms to dwell and live, while soil organisms provide the organic binding agents responsible for soil structural development and stability. Soil structure is not static but changes dynamically in response to spatial and temporal changes of soil biota, climate, and management.

20.8.6 Soil Organic Carbon Content

Climate change is expected to reduce soil organic C content and increase losses of CO_2. The soil organic C storage decreases with increase in temperature and increases with increase in soil water content (Fig. 20.5). Increase in soil temperature accelerates decomposition of crop residues. Some estimates show that a 3°C increase in temperature is projected to decrease soil organic C concentration by about 11% in the upper 30-cm soil depth and increase CO_2 emissions by 8% (Buol et al., 1990). Initial rates of CO_2 emissions in response to an abrupt elevation in soil temperature can be particularly high (Bergh and Linder, 1999; Melillo et al., 2002).

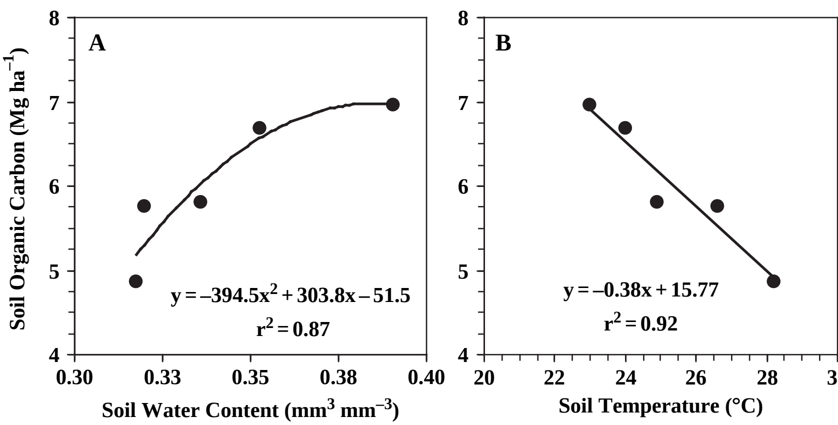

Fig. 20.5 Organic C pool increases with increase in water storage capacity and decrease in soil temperature in the 0- to 2-cm depth of no-till soils (After Blanco-Canqui and Lal, 2007)

20.8.6.1 Labile and Stable Organic Materials

There are two fractions of soil organic C pool: active/labile and passive/stable. The active fraction is more sensitive to temperature increase and is more easily decomposed than the passive fraction. The passive fraction consists of aromatic compounds of complex structure while the labile fraction consists mostly of polysaccharides. The ephemeral emissions of CO_2 result mainly from the labile fraction decomposition. Preferential loss of labile organic C pools under climate change reduces soil C pools.

20.8.6.2 Effects of Land Use Change

Changes in land use and management influence losses and gains of soil organic C. The switch of land from C sink to C source can alter the overall terrestrial C cycle. Deforestation, intensive tillage, and crop residue removal are the greatest causes for soil organic C losses and global warming. Adverse effects of global warming on soil organic C storage may decrease over time as the system is acclimated as a function of soil type, management, and ecosystems characteristics. On a mixed hardwood forest in a temperate region, an increase in soil temperature by 5 °C by artificial heating accelerated decomposition of soil organic matter and increased CO_2 fluxes by 28% in the first 6 yr, but the warming effects in the last 4 yr of the 10-yr study diminished, showing that the initial losses in C were transient (Melillo et al., 2002).

20.8.6.3 Effects of Climatic Regions

The higher soil organic matter decomposition rates may be, however, offset by increase in precipitation in temperate and humid regions. Decomposition of organic matter under climate change may be enhanced in cool and wet climates due to soil warming where current decomposition rates are slow. Changes in soil organic C content due to soil warming are projected to be region-specific. Rates of organic matter decomposition and accumulation can also vary among soils. Fluctuations in soil temperature may have greater effect on soil organic C dynamics in poorly-structured soils. Increases in soil temperature and decreases in soil water content due to climate change may have slower effects on stable and clayey soils than in sandy or sloping erodible soils with limited aggregation. Indeed, a systematic removal of corn stover mulch from sloping and erodible silt loam soils increased mean daily soil temperature and reduced soil water storage resulting in rapid reductions in C content, but had no significant effects on a flat and clayey soil in a temperate region (Blanco-Canqui and Lal, 2007).

20.9 Crop Production

The projected climate change may have strong impact on cultivated lands, pasturelands, and forestlands. Climatic and crop models show that agricultural crops are particularly sensitive to changes in CO_2 concentration, temperature, solar radiation, and precipitation (Southworth et al., 2002a). Climate change may affect planting dates, time to maturity, harvesting dates, crop yields, and thus the farm operation and the agricultural economy.

20.9.1 Positive Impacts

Increase in rainfall, temperature, and CO_2 concentration can boost crop production (Fig. 20.6). It can increase the length of cropping season and allow completion of two cropping seasons in one year. Greater CO_2 concentrations can also improve water use efficiency by closing stomates while still enhancing photosynthesis (Bergh and Linder, 1999). It is estimated that soybean yield may increase by an average of 40% and wheat yield by about 20% under the new climate conditions between 2050

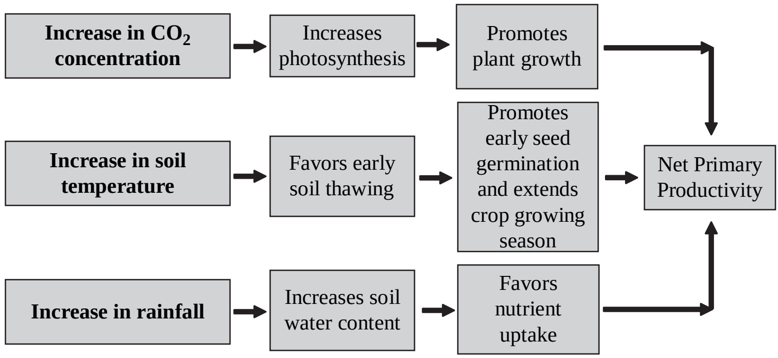

Fig. 20.6 Beneficial effects of climate change on biomass production

and 2059 (Southworth et al., 2002a, 2002b). A 10% increase in CO_2 emissions may increase wheat yield by 5% in the Midwest USA (Zhang and Nearing, 2005).

Elevated soil temperature will likely increase mineralization of N and activity of microbial activity, thereby releasing nutrients to soil. The available N would increase plant growth and C storage in the tissues, which can offset or at least partially compensate for the C losses from the soil. The C storage (1560 gm^{-2}) in hardwood trees as a result of increased N release by artificially raising the soil temperature by 5 $^{\circ}$C equaled the losses from soil over a 10-yr period in a cool temperate forest (Melillo et al., 2002). Since elevated soil temperature increases organic matter decomposition, production of plant materials with high C:N ratio may reduce decomposition and promote gains in soil C.

20.9.2 Adverse Impacts

While global warming may promote plant growth at mid and high latitudes (e.g., USA, Alaska, Canada, Russia), it may, however, have adverse effects on arid, semiarid, and tropical regions (e.g., Amazon). Plants and soil organisms in higher latitudes are already adapted to fluctuations in temperature and rainfall, whereas those in tropical regions may be more sensitive to climate changes. A small shift in temperature may have little effect on higher latitudes, while the same shift may have large effects on lower latitudes. In the USA, crop yields in the northern states may increase whereas those in the southern states may decrease under the new climate scenario (Southworth et al., 2002a).

The beneficial effects of high CO_2 concentrations on biomass production may be offset by increasing air temperature and reduced precipitation in dry regions (Fig. 20.7). Increased soil temperature can accelerate evaporation and decrease water storage, thereby decreasing crop production. Increased temperature can also hasten maturity of agricultural crops and reduce yields. Demands for irrigation water and competition for water resources may increase in arid and semiarid regions.

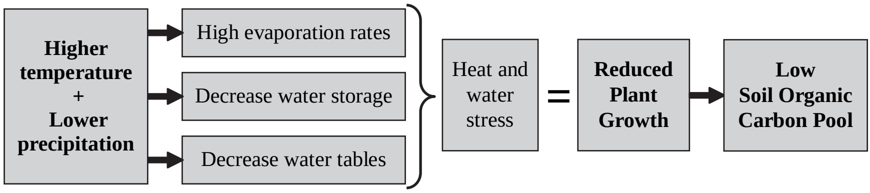

Fig. 20.7 Adverse influence of climate change on crop production

Enhanced vegetation growth through higher CO_2 emissions and soil water content may also deplete nutrients and eventually reduce plant growth. Soils poor in fertility could be impacted more by increased nutrient extraction. Additional application of fertilizers may be needed to compensate for the high nutrient extractions. Greater mineralization of soil organic matter may cause N losses. Higher temperatures may promote proliferation of insects and pests in regions which are currently cold.

20.9.3 Complex Interactions

Increasing air temperature may either increase or decrease biomass production depending on other plant growth factors (e.g., rainfall, CO_2 emissions). Vegetation growth not only depends on air temperature and CO_2 concentration in the atmosphere but also on plant available water and availability of essential nutrients. The combination of all these factors influences vegetation growth. Rate of CO_2 absorption also varies with crop type and age. The C_3 plants (e.g., wheat, rice, soybeans) are more sensitive to changes in CO_2 concentration than C_4 plants (e.g., corn, sorghum, sugarcane).

20.10 Soil Warming Simulation Studies

Impacts of the projected global warming on soil properties, soil organic C, and soil erosion have been simulated by warming soil plots. Various methods of artificial soil warming are available:

20.10.1 Buried Electric Cables

This is a common method to warm up the soil in small plots (Melillo et al., 2002). This method consists of burying parallel electric cables into the soil spaced 10 or 20 cm apart at a specified depth (e.g., 10 cm). A difference of temperature (e.g., 5 $^\circ$ C, 10 $^\circ$ C) between the heated and control plots is maintained throughout the simulations. Temperature in the plots is controlled with an automatic datalogger that switches off when the temperature is above the difference and switches on when temperature drops below the difference in order to maintain soil temperature within of $\pm 0.1^\circ$ C. A network of thermistors is used to monitor the soil temperature in the plots.

20.10.2 Overhead Heaters

This method uses electric heaters which are suspended above the experimental plots to a specified distance (e.g., 2 m, 3 m) (Harte et al., 1995). Heating uniformity of the plot is ensured by means of radiometry and soil temperature measurements. It requires well-designed reflectors to reduce propagation of heat outside the plots and ensure a uniform irradiation of heated plots. The visible artificial light from the heater must also be designed in a way that does not interference with photosynthesis nor the suspended heaters block the sunlight.

20.11 Modeling Impacts of Climate Change

Up to now, studies on the impacts of the projected climate change on soil erosion, C dynamics, and crop production have mostly relied on climatic models. Various scenarios of climate change, soil hydrologic processes, vegetative cover, and management for different ecosystems are constructed and simulated in their impacts on soil erosion. Models have extrapolated data from small-scale plot or lab studies to landscape or watershed level. Among the climate change models used are LISEM, USLE, RUSLE, KINEROS (Kinematic Runoff and Erosion model), SWAT, and WEPP (Nearing et al., 2005; Zhang, 2006).

Models can be used to simulate the following changes that could directly affect soil erosion:

- Different levels of atmospheric CO_2 concentration
- Increase/decrease in precipitation and changes in rainfall intensity and frequency
- Abrupt or gradual increase in air and soil temperatures
- Different types of vegetative cover (e.g., shifts in land use and management systems
- Changes in evapotranspiration rates, soil water storage, and drought periods
- Changes in soil erodibility as affected by changes in SOM and surface properties
- Shift in cropping systems due to changes in cropping seasons, planting dates, and market prices
- Shift of crops to different types of soils and topography and impact of heat and water stress

The WEPP is one of the most widely utilized models for predicting soil erosion under new climate changes because it permits the simulation of complex and dynamic processes of rainfall and temperature interactions. It can simulate residue decomposition rates by integrating effects of changing soil temperature, water content, and microbial activity as affected by the new climate. Rainwater infiltration, soil compaction, evapotranspiration rates, canopy and residue cover, surface roughness, and residue decomposition rates are major determinants of runoff and soil loss in the WEPP model. Modeled results all point toward an increase in runoff and soil erosion rates under climate change with a magnitude depending on local and regional conditions.

20.12 Adapting to Global Warming

Impacts of climate change on soil and water conservation are expected to be more negative than positive. Increases in water and wind erosion are already happening now (SWSC, 2003). Unless aggressive changes in conservation strategies are made, the projected climate change will reduce soil productivity, increase risks of water and wind erosion as well as water pollution. Current soil conservation practices are designed and managed based on past climatic data rather than on new climate conditions. Thus, improved conservation practices must be engineered based on the projected climate change data. Current conservation practices may not be effective enough to withstand projected intense precipitation and high volumes of runoff. Most soil erosion occurs during infrequent but very intense rainstorms (Larson et al., 1997). Continually adapting conservation practices to climatic changes must be designed to reduce greenhouse gases, sustain crop production, and reduce water pollution.

Prudent soil management is crucial to ecologically minimize global warming. The best strategy is to put back the C that has been lost. Practices that take up CO_2 from the atmosphere and promote the terrestrial C sink in the soil must be implemented. Among these practices are reduced tillage, no-till, manuring, residue return after harvest, and planting of tree and grass. A soil becomes a potential source of CO_2 emissions if trees are cut, intensively plowed, and residue after harvest removed. Thus, reforestation of cleared lands and afforestation of marginal and degraded lands could enhance CO_2 uptake. Between 40×10^9 and 90×10^9 Mg of C has been lost since the inception of intensive agriculture (Reay, 2007).

No-till farming with residue mulch in rotation with cover crops is a viable alternative for sequestering soil organic C and offsetting the net emissions of CO_2 (Lal, 2007). Predictive models indicate that no-till management is the best strategy to reduce increases in soil erosion under the projected climate change (Zhang and Nearing, 2005). It reduces wind erosion in dry regions by conserving water and reducing evaporation rates. As of right now, no-till is mostly practiced in USA, Brazil, Canada, South America, and Australia. The practice of no-till farming by small-scale and resource-poor farmers in developing regions (e.g., sub-Saharan Africa, South Asia) where erosion rates are the highest is practically negligible (Lal, 2007). Coincidentally, these are also the regions where climate change is predicted to have the harshest effect (IPCC, 2007). Thus, strategies for no-till adoption and development of related conservation effective practices are needed to adapt to climate change in erosion prone environments. In developing countries, resource-poor farmers utilize crop residues as animal fodder and fuel source. Thus, no-till management without residue return is as counterproductive as plow tillage for soil erosion control. Residue management may become more important under changing climate. Benefits of no-till to C sequestration are discussed in Chapter 17.

Among other strategies are establishment of cover crops, bioenergy crops, crop rotations with N-fixing legume forages, and short-rotation plantations of trees for fuel, food, and wood. Practices that maintain permanent surface cover and supply organic matter improve the soil resilience. Reduction of excessive grazing and wheel traffic are related strategies to reduce soil compaction and erosion. Conservation buffers such as grass barriers and vegetative filter strips established at the bottom of croplands trap and degrade sediment and nutrients while reducing non-point pollution. No single conservation practice is adaptable to all soil conditions, thus site-specific management strategies for each ecosystem and soil must be designed to counteract potential adverse impacts of climate change.

Summary

The projected global climate change may exacerbate soil erosion risks through the alterations in precipitation and temperature patterns. The consistent increase in ocean temperatures, sea level, melting of ice and snow from glaciers and snow capped mountains, flood frequencies, wind storms, tropical cyclones, and shifts in temperature and rainfall patterns are signs of climate change, which can undermine land stability and alter erosion dynamics. Runoff rates are expected to increase by 30 to 40% in high latitudes, while wind erosion would increase in arid and semiarid regions.

The magnitude of increases in water and wind erosion due to climate change is most likely to be region-specific and dependent on the ecosystem, topography, and management. Adverse impacts of climate change on soil erosion are expected to be the greatest in developing regions of the world with degraded soils and limited accessibility to effective erosion control measures. Higher temperature and lower precipitation rates may increase wind erosion in arid and semiarid regions. At present, about 25% of cultivated lands in arid and semiarid regions are already severely affected by wind erosion. Increase in soil erosion due to the new climate can reduce grain and biomass production. Reduced vegetative cover and biomass input can increase erosion and cause further soil degradation and desertification.

Field experimentation and modeling are two approaches used to understand the potential impacts of new climate on soil erosion. Buried electric cables and overhead electric heaters are used to warm up soil in small plots to simulate global warming. Response of soil attributes and crop production to artificial warming is then monitored over time. Models allow the simulation of various scenarios of climate change (e.g., fluctuations in temperature and rainfall), different levels of atmospheric CO_2

concentration, soil erodibility, vegetative cover, tillage and cropping systems, and soil management.

Strategies against global warming include the establishment of soil conservation practices such as no-till technology, crop rotations, cover crops, bioenergy crops, N-fixing legume forages and trees, and short-rotation plantations of trees for fuel, food, and wood, conservation buffers. Any practice that maintains permanent surface cover may decrease fluctuations in temperature and reduce adverse impacts of rainfall on soil erosion. Performance of conservation practices may change under the new climate. Thus, the current conservation practice must be acclimated and new strategies developed.

Study Questions

1. Differentiate between greenhouse effect and global warming? Discuss causes and mechanisms of global warming.
2. Discuss the types of greenhouse gases responsible for global climate change.
3. Describe greenhouse gases based on their magnitude of contribution to global warming.
4. Explain the mechanisms by which global climate change may affect soil erosion.
5. Explain how a decrease in rainfall in arid regions could lead to higher rates of soil erosion.
6. Discuss the impact of global climate change on factors and processes of soil erosion.
7. Recommend soil management strategies to offset the projected abrupt changes in precipitation and temperature.
8. Describe the magnitude of glacier melting in the world and its impacts on soil erosion.
9. Discuss how climate would increase or reduce biomass production.
10. Discuss relationships between soil formation and climate change and their influence on soil erosion.

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Chapter 21
The Way Forward

Two among major issues of the 21st century are global food insecurity and environmental degradation. Rapid growth of human population has increased concerns of soil degradation (e.g., soil erosion, low soil fertility) and food insecurity. The world population is projected to increase by about 2.6 billion between 2000 and 2050 (UNIS, 2005). The rapid economic development in some parts of the world (e.g., China, India) is expected to increase the pressure on land and also aggravate environmental quality. The ongoing problems of soil and environmental degradation may most likely become a greater concern in the near future.

Despite the advanced technology and modern science, soil degradation and food insecurity problems persist. To date, billions of dollars have been spent annually on soil and water conservation programs particularly in industrialized countries, but environmental problems such as non-point source pollution of water and hypoxia remain to be major issues. Installation of costly erosion control structures (e.g., terraces, diversion channels, sediment basins) and adoption of conservation tillage (e.g., reduced tillage, no-till) have not completely alleviated the problem of non-point source pollution. Some technologies have created serious environmental (e.g., excessive use of fertilizers and pesticides) challenges. Heavy machines used in mechanized farm operations lead to drastic soil disturbance which necessitate use of proper technologies to conserve soil and water and restore soil structure. Terraces and other engineering structures for erosion control used in industrialized countries are often too expensive for resource-poor farmers in developing countries.

Soil functions are becoming more diverse (see Chapter 1), and so are the needs of its conservation and management. There is a need for paradigm shift and a new direction in soil conservation and management. Effectively addressing the contemporary problems requires the formulation of new policies, development of new technologies, improvement of current practices of soil and water conservation, and the adoption and implementation of conservation effective measures. Prudent soil conservation and management must be a high priority at a time when demands on soil resources have become more critical than ever before. Lessons learned from historic failure and successes in soil and water conservation are guiding principles to addressing the emerging challenges with a new vision.

Soil conservation efforts have had some important successes, yet more remains to be done. This is particularly true in poor regions of the world where deforestation,

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slash-and-burn agriculture, and mismanagement are caused by helplessness and desperate attempts to survive. Therefore, the importance of conserving and restoring productivity of soil and quality of water resources in a sound manner can not be overemphasized. More and more food must be produced from less and less land. Unless proper measures to improve soil fertility by conserving soil and water are established, soil degradation may accelerate and adversely affect the long-term productivity and sustainability.

Humankind has the greatest responsibility to reverse soil degradation trends and ensure the food security for the present and future generations. There is a need for high level commitment in all sectors of society to protect the soil and environment and maintain/improve the soil's productivity to meet the ever increasing demands. In some parts of the world (e.g., Africa), agricultural soils have reached an alarming level of degradation (e.g., low fertility, severe erosion) which demands an urgent attention to reduce risks of famine and social and ethnic conflicts.

This concluding Chapter outlines some important strategies to conserve soil and water resources based on the lessons learned and the future needs for soil conservation and restoration.

21.1 Strategies of Soil and Water Conservation

The problems of soil and water conservation must be addressed through a concerted effort of soil stewardship and technological input. A prudent approach to soil and water conservation requires a holistic approach to solve practical problems that affect not only farmers but the entire society. An effective soil and water conservation program is linked to political, social, and economic conditions of each region. Some of the strategies are the following:

1. Development of economic and conservation-effective practices to: (1) restore degraded soils, and (2) maintain and enhance the productivity of prime agricultural soils.
2. Identification and development of site-specific conservation practices based on local and regional biophysical, social, cultural, and political forces. There is no panacea, and no single practice fits all situations.
3. Establishment of pilot programs for on-farm demonstrations of improved soil and water conservation practices based on a multidisciplinary and farmer participatory approach in regions with resource-poor farmers (e.g., Africa, South Asia, South America) (Lal, 2007).
4. Establishment of programs that reward farmers for their commitment to soil stewardship as well as for the successful implementation of strategic conservation practices such as management of soil erosion, conservation of water in the root zone, reduction of soil compaction, alleviation of crusting and surface sealing, improvement in soil fertility, and installation of water ponds to harvest and recycle rainwater.

5. Installation of conservation-effective practices which keep the soil in place and reduce both the on-site and off-site effects of soil erosion. Conservation practices which minimize soil detachment and transport and reduce runoff rate and amount must be developed and refined. Erosion control practices trap sediment and chemicals at the downslope end of source areas or just above the water sources, while conservation practices keep the soil in place. Sediments deposited in the footslopes at the lower end of fields are considered a lost soil because it can neither be easily nor economically brought back to its original location.
6. Development of technologies to alleviate shortcomings of the conservation tillage systems (e.g., reduced tillage, no-till) such as low crop yields, excessive use of herbicides and fertilizers, stratification of soil organic C and nutrients near the soil surface, and interference of residue mulch with planting operations and soil warming.
7. Development of conservation practices which reduce both the water pollution and emission of greenhouse gases. Treatment of polluted water is expensive and degradation of water quality difficult to rectify. Thus, runoff of water and pollutants must be minimized by improving infiltration and water retention capacity of the soil.
8. Widespread adoption of soil conservation practices such as growing cover crops, planting of N-fixing trees, maintaining riparian buffers, establishing field borders, adopting strip cropping, using crop residue mulches, growing green manure crops, using agroforestry, and other biological measures in combination with mechanical structures (e.g., terraces).
9. Conversion of severely eroded soils to restorative land use such as perennial vegetation covers, and restoration of degraded and marginal soils through improvement in soil organic matter reserves and creation of a positive nutrient, C and elemental budget.
10. Refinement of the threshold levels of soil erosion or T values for each soil and region based on research data.
11. Increased emphasis on research and education along with the transfer of conservation effective technology to land managers and stake holders by strengthening networks and improving connectivity.

Needs for additional basic and site-specific research include the following:

1. Addressing site-specific priorities in soil and water conservation research.
2. Collecting on-farm data on water, wind and tillage erosion and their impacts on soil productivity with emphasis in developing countries.
3. Collating data on erosion impacts on crop yields under on-farm conditions. Credible data are needed to obtain reliable estimates of the magnitude of the problem for creating awareness of policymakers, farmers, and the general public about the negative impacts of erosion on food security and environmental quality.
4. Conducting basic research on soil erosion process particularly in transport and deposition processes in relation to soil C dynamics, climate change, and emissions

in greenhouse gases. Greater understanding of soil erosion processes would lead to development of appropriate techniques of control and improvement in soil, air, and water resources.
5. Collating and synthesizing data on the performance of soil conservation practices and magnitude of soil erosion before and after adoption of control practices.

21.2 Soil Conservation is a Multidisciplinary Issue

Soil conservation is a wider and all encompassing problem than hitherto perceived. It is a discipline intrinsically linked to numerous disciplines in natural sciences (e.g., geology, hydrology, climatology, engineering) and social sciences (e.g., economics, policy, human dimension). Thus, understanding processes of soil erosion requires the knowledge of basic principles of pedology, forestry, agronomy, climatology, sedimentology, hydrology, engineering structures, and other fields of soil-plant-atmosphere relationships. Soil conservation does not only involve control of runoff and soil erosion but also comprises a wide range of management and practices including irrigation, application of amendments, fertilization, and drainage aimed at increasing and/or maintaining soil productivity. A successful program in soil conservation also requires active involvement of land owners, farmers, economists, social scientists, policymakers, and the general public. A greater interdisciplinary effort is necessary for developing soil conservation technology and adoption of these innovations at the local, regional, and national levels.

21.3 Policy Imperatives

Availability of funds for conducting research on soil and water conservation is dwindling. Government agencies and other institutions across the world must give a high priority to this issue and increase support to soil erosion research for generation of new technologies and strategies. Conservation policies must be directed to mitigating social costs and improving the economic conditions of farmers. Among the policies needed for soil and water conservation include the following:

1. Providing more funding for a strong enforcement of soil and water conservation programs.
2. Providing training and financial incentives for good soil stewardship.
3. Improving accessibility and providing more opportunities for education about the benefits of soil conservation and stewardship.
4. Giving technical and financial assistance for establishing soil conservation practices.
5. Emphasizing and prioritizing research and development of new technologies.
6. Developing regulations or measures for the conversion of degraded croplands to a permanent vegetation cover.

7. Providing support for exchange of information among farmers, extensionists, and other institutions.
8. Supporting active and participatory research, extension, and training programs in developing countries with predominantly resource-poor farmers.

21.4 Specific Strategies

Specific strategies must be identified and implemented for effectively addressing global issues of the 21st century. Some of these strategies are discussed below:

21.5 Food Production

The surge in food production in industrialized nations since the World War II has not been replicated across the world. Food insecurity remains to be an increasing concern among developing countries in sub-Saharan Africa and South Asia. Agricultural production in these regions must be doubled by 2050 in order to meet the growing demands for food (Wild, 2003). While there is abundant food production on a global basis, the food produced is not accessible to those who need it. Above all, the relationship between soil erosion and crop production is poorly understood and requires the following:

1. Accurate estimation of the impacts of soil erosion on crop/animal production at different scales.
2. Comprehensive economic analysis of erosion impacts by crop, soil order, biomes, nation, and continent (den Biggelaar et al., 2001).
3. Development of mathematical models to assess soil erosion- crop yield relationships for different crops, soils, climates (Todorovic and Gani, 1987).
4. Development of GIS-based models designed specifically to assessing crop yield-erosion patterns across a broad range of geographical regions.
5. Quantification of rates of erosion vis-à-vis and amount of crop production to understand the magnitude of soil erosion.

Soil restoration and crop production in developing countries may be enhanced by:

6. Reducing and reversing soil degradation trends and restoring degraded/desertified soils with adoption of improved conservation practices.
7. Managing eroded soils and landscapes with the rational use of chemical fertilizers and increased use of organic amendments through strategies of integrated nutrient management.
8. Improving access to fertilizers for the resource-poor farmers.
9. Identifying and improving new crop varieties adapted to eroded soils and tolerant to biotic and abiotic stresses.
10. Designing and improving rainwater harvesting techniques for supplemental irrigation in arid and semiarid regions.

11. Promoting the adoption of no-till farming with N-fixing cover crops and diverse rotations.
12. Promoting crop residue retention on soil and using fertigation techniques to provide nutrients by appropriate formulations which minimize losses and increase use efficiency.

21.6 Crop Residues and Biofuel Production

An emerging need is the production of large amounts of biomass for producing renewable energy. Crop residues are perceived as the top candidates for biofuel feedstocks. Crop residues are not a waste and their use for producing ethanol comes with a heavy price (Blanco-Canqui and Lal, 2007). Increasing areas for producing corn in the USA and sugarcane in Brazil and other parts of the world can aggravate the problem of water runoff and soil erosion. The need for producing larger amounts of biomass for biofuel can add pressure on scarce resources and alter soil and water conservation strategies. Harvesting all the biomass produced off the agricultural fields can exacerbate soil erosion and increase water pollution. The implications of residue harvesting must be adequately and accurately characterized by:

1. Assessing the impacts of crop residue removal on soil properties, soil erosion risks, crop production, and environmental quality for diverse soils and regions.
2. Managing crop residues for a number of competing uses such as soil and water conservation, animal feed, biofuel feedstocks, and industrial raw materials.
3. Balancing the residue requirements for conserving soil and water with those for satisfying energy demands by evolving an optimum plan of residue management. Such a plan would benefit domestic energy production while minimizing negative impacts on soil organic C storage, agricultural production, and environmental quality.
4. Evaluating the relationships among soil type, management, tillage, climate on the amount of biomass-C input to maintaining soil organic C pool and sustaining crop productivity and environmental quality.
5. Establishing soil-specific recommendation and guidelines of permissible levels of crop residue removal based on the site-specific needs for soil and water conservation, soil organic C storage, soil productivity, and environmental quality. Information on the threshold levels is needed to support energy industries in providing decision support system.
6. Intensifying research on bioenergy plantations based on dedicated crops such as warm season grasses, short-rotation woody perennials, and mixture of prairie grasses as alternatives to crop residue removal as biofuel feedstocks.
7. Modeling of short-, medium-, and long-term impacts of crop residue removal on soil productivity, ecosystem services, and agronomic productivity. Specific models for crop residues and biofuel production are not available. A rigorous modeling of crop residue management implications on soil and water conservation is warranted using site-specific information.

21.7 Biological Practices and Soil Conditioners

Much remains to be researched about the biological and agronomic measures of soil erosion control. Some of the research priorities include the following:

1. Promoting adoption of cover crops and other biological measures for reducing soil erosion and enhancing C sequestration across a wide range of ecosystems.
2. Assessing the performance of cover crops based on tillage systems, soil type, and climate by collecting site-specific experimental data across principal soils and ecoregions.
3. Demonstrating the on-farm benefits of cover crops for small-scale farmers particularly in developing countries.
4. Determining the optimum application rates of soil conditioners such as polyacrylamide, nanoenhanced compounds, and zeolites on watershed scale for different soil and climate conditions.
5. Using biotechnology and information technology in conjunction with nanotechnology to addressing the age old problem of soil erosion and sedimentation.

21.8 Buffer Strips

Buffer strips are biological measures of soil conservation and restoration. Specific needs for research and development include the following:

1. Demonstrating the benefits of the systems to control soil erosion and improve agricultural production and overall environmental quality across diverse agroecosystems and climate conditions. Despite the numerous benefits of buffers to soil and water conservation, adoption of the practices at large-scale across agricultural systems in the USA and around the world has been slow. Performance of buffer systems often differs with management, vegetation specie, soil type, and topographic conditions.
2. Strengthening the database on the water quality effects of buffer management. In principle, buffers can retain significant amounts of sediment and chemicals leaving upstream ecosystems, but there is little information about the actual quantitative impact of buffers on water quality for human and animal consumption.
3. Extending the research domain on buffers from site-specific studies to a watershed scale. Field scale studies can provide more representative data as well as broaden the understanding of the potential of buffer strips for reducing risks of water pollution. Most of the studies on the different types of buffer strips have been conducted on small site-specific plots.
4. Understanding the temporal and spatial performance of buffer strips. Riparian buffers and filter strips may fail to control concentrated flow erosion. Effectiveness of buffer strips in reducing the transport of pollutants for different runoff flow types and landscape topographic conditions must be documented.

5. Enhancing the understanding of the effectiveness of stiff-grass barriers for improving water quality in large field-scale studies. While riparian buffers, grass filter strips, and grass waterways have been widely studied, effectiveness of vegetative barriers established by native and warm season grasses for reducing sediment, N, and P losses in runoff is not well understood.
6. Improving information on the performance of multi-species buffer systems using a combination of sod-forming with bunch-type grasses or warm season and native grasses practices for reducing transport of sediment and non-point source pollutants in concentrated flow and preventing formation of ephemeral gullies.
7. Developing, calibrating, and validating process-based models and pedotransfer functions to estimate the effectiveness of buffer strips. Most of the models for sediment transport prediction were designed for sheet or interrill flow conditions and not for concentrated flow. Models may not be applicable when the buffer areas adjoining the upstream lands are inundated or filled with sediment under concentrated flow.
8. Using field scale modeling to extrapolate data from representative studies to large watersheds or actual field conditions. Improvement/ validation of current models and development of new models are high priorities on buffer strip research. Computer based models such as VFSMOD, RUSLE-2, WEPP, and SWAT including the simplified equations can be used to: (1) understand the intrinsic mechanisms of buffer functions for soil erosion control, (2) evaluate pollutant trapping efficiency of various-sized buffers, and (3) determine the optimum size of buffers. Models can help with estimating future trends in water pollution control as well as economic benefits under different scenarios of buffer management.
9. Refining and expanding field scale, conceptual, continuous time, and buffer-specific models to assess the impact of management and climate on buffer performance for controlling non-point source pollution in large watersheds and water bodies across diverse ecosystems.

21.9 Agroforestry

Soil and site-specific research is needed to assess effectiveness and application of agroforestry with regards to the following:

1. Identifying tree domestication strategies, selecting proper tree species or taxa based on their products or services, farmer- and market-based tree species, and developing management guidelines for pest control, commercialization of products, design of optimum spacing and width of hedgerows and alleys, creation of germplasm banks of plant materials, and tree breeding.
2. Assessing above- and below-ground tree-crop interactions and limiting growth factors in space and time that govern the effectiveness of agroforestry practices for conserving soil and water and improving crop production and economic returns.
3. Conducting more region-specific assessment of positive and negative aspects of forest farming (e.g. fiber farming) on soil erosion and water quality to develop

ecologically and economically sustainable forest management strategies to conserve soil and water resources.

4. Studying dynamics of vadose zone hydrology in agroforestry systems responsible for surface and subsurface runoff, lateral flow, and nutrient and pesticide leaching, and groundwater pollution. Nutrient dynamics and possible leaching of mobile chemicals below the coarse root zone of trees have important implications to establishing permanent and long-term hedgerows of trees in agroforestry systems.
5. Collating and synthesizing quantitative information about sediment and nutrient filtering capacity of agroforestry, forest farming, and silvopasture systems to provide guidance in designing effective agroforestry practices.
6. Evaluating soil properties in relation to tree species accounting for the differences in above- and belowground biomass input, patterns of root systems, tree height, and canopy cover.
7. Assessing soil physical and hydrological properties at different distances from the hedgerows in alley cropping systems. Such information is critical to understanding the magnitude of changes in soil attributes and their effects on alley crops as well as for the expansion of alley cropping systems.
8. Determining ability of agroforestry systems to improving crop production in marginal or degraded soils in response to the ongoing expansion of agricultural production.
9. Developing protocols for accurate assessment of C pools and fluxes of greenhouse gases. It is hypothesized that current methods of C assessment may overestimate the potential of agroforestry systems for C sequestration. Furthermore, measurements of C stocks have constraints in regards to sampling and methodology of determination.
10. Assessing potential of C sequestration and reducing emissions of greenhouse gases by agroforestry systems across regions and their implications for controlling global climate change with emphasis in forest farming and silvopasture. The real potential of agroforestry for reducing greenhouse gases such as CO_2, N_2O and CH_4 emissions is not well known. Yet, these data are needed to develop decision support systems to promote/improve these agroforestry systems for mitigating global climate change.

21.10 Tillage Erosion

The emerging scientific field of tillage erosion needs additional research with regards to the following:

1. Increasing research on tillage erosion to understand the basic processes. Most of the discussions on soil erosion research are still concentrated on water and wind erosion. The reduced emphasis on tillage erosion has limited the design of management strategies across a wide range of tillage scenarios for different soil and climate conditions.

2. Generating credible data on the rates and magnitude of tillage erosion across a range of soils with contrasting slopes and management systems for developing decision support systems to manage tillage erosion. The risks of past, present, and projected tillage erosion should be assessed and mapped for major soils.
3. Understanding the magnitude of changes in soil properties (compaction parameters, soil texture, water and organic matter content, coarse fragments) due to tillage erosion and relating the changes to spatial variation in crop yields. Understanding of the impacts of tillage erosion on crop production and soil productivity is fragmented.
4. Developing an innovative theoretical model to predict soil transport by tillage for a range of tillage implements, management scenarios, soils, landscape positions, and ecosystems. Models that simultaneously simulate tillage, water, and wind erosion and partition the various sources of erosion are needed for an accurate estimation of total erosion in complex terrains. Improved models are also needed to assess tillage erosion across contrasting topographic positions with complex slope gradients and morphologies, reflecting the natural landscape domains.
5. Conducting experiments to properly calibrate and validate existing models by ground truthing. Data from long-term field experiments are needed to understand the magnitude of tillage erosion and develop a physically-based model capable of simulating tillage erosion implications across ecosystems.
6. Expanding research to real-world situations where large fields rather than small plots undergo repetitive tillage operations across seasons under different levels of soil water content.
7. Developing techniques and models to separate soil losses by tillage erosion from those by water and wind erosion. Lack of standard techniques to partition the different components of soil erosion limits the accurate estimation of the magnitude of tillage erosion.
8. Assessing and understanding indirect and direct effects of tillage-induced soil erosion on C sequestration, emissions of greenhouse gases, and nutrient cycling. Buried soils in foot- and toe-slopes may have a large sink capacity for increasing long-term C sequestration. Elucidation of sediment-borne organic matter is a sink or source of CO_2 is a research priority.
9. Assessing a detailed spatial and temporal magnitude of tillage erosion to fully understand the severity of losses across different ecosystems, soil conditions, and climate. Optimum harvest dates for root crop harvesting needs to be developed based on solid decision support systems that account for the changes in the various factors affecting soil losses by harvest.

21.11 Organic Farming

Organic farming is an ecological alternative to conventional farming, but harnessing benefits of this option necessitate the following research:

1. Managing effectively and economically weeds, pests, and disease infestation.
2. Adapting biological control practices including crop rotations, cover crops, pest and insect habitat management, and other techniques.
3. Collating data on the impacts of organic farming for reducing soil erosion, improving soil physical and chemical quality and increasing crop yields.
4. Developing farmer-oriented and applied research to collate comprehensive data on the long-term benefits of organic farming.
5. Developing a framework to evaluate production costs of organic farming and its implications to crop yields and soil and water quality.
6. Conducting long-term trials on organic farming adjacent to conventional farming systems to compare differences across a wide range of soils and management scenarios.
7. Strengthening support services including extension agents or trainers to provide information on organic farming practices to farmers.

Proper management of organic farming is crucial to obtaining the desired benefits. Some recommendations to achieve these benefits include the following:

1. Reducing the excessive application of manure and improper storage of organic amendments.
2. Decreasing the need for performing frequent (e.g., annual) soil tests to determine the amount of organic fertilizers to be applied.
3. Preparing compost mixtures of crop residues, animal manure, and other organic materials to obtain a balanced organic fertilizer.
4. Using crop rotations and cover crops as integral components to combat diseases, control weeds, and recycle nutrients.
5. Avoiding application of organic fertilizers just before irrigation or on saturated soil, which increases risks of nutrient loss and water pollution. Timing of application is important to reducing losses of nutrients.
6. Combining organic farming with other conservation practices besides rotations and cover crops (e.g., no-till, reduced tillage, grass barriers, vegetative filter strips, riparian buffers).
7. Protecting manure and compost piles to reduce runoff and contamination of water.
8. Reducing the rate of application of fresh manure or compost which may contain pathogens.
9. Analyzing manure used as fertilizer and test the soil regularly for changes in the accumulation of these heavy metals (e.g., lead, iron, cadmium, arsenic) and other contaminants.

21.12 Soil Quality and Resilience

The following challenges must be addressed with regards to the concepts, methods of assessment, and evaluation criteria of soil quality and resilience:

1. ***Definition.*** Current definitions of soil quality and resilience are subjective and not operationally suitable for all soil environments and management conditions. A need exists to develop a scientific concept of soil quality and resilience. The new concept should be comprehensive and validated for different uses and functions of the soil.
2. ***Assessment methods.*** Development and refinement of standard and unique quantitative methods are needed for measuring parameters of soil quality and resilience. Research in soil quality and resilience can benefit from the use of precision agriculture technology, statistical tools (e.g., autocorrelograms, semivariograms, kriging), and other modern research tools. The present challenge is to develop a sophisticated framework of soil quality and resilience assessment based on strong scientific principles. Soil quality indexing may be more acceptable by the scientific and farm community if site-specific and purpose-oriented indexing is done based on solid scientific parameters of evaluation using soil taxonomy as the basis for understanding soil functions.
3. ***Indicators.*** A unique set of relevant and sensitive indicators/parameters for the evaluation of soil quality and resilience for different soils and ecosystems must be identified. Attributes controlling the soil resilience must be classified and incorporated into the official survey maps and reports for each soil. Current selection of soil quality indicators, normally based on soil's suitability for crop production, require modifications to be sensitive enough to other soil functions (e.g. water quality, C sequestration, reduction in net greenhouse gas emissions). Identifying valid indicators of soil quality requires the integration of inherent and dynamic soil properties and their interactions with time and space.
4. ***Long-term experimentation.*** Significance and agronomic impacts of soil quality and resilience can only be understood through the long-term experimentation for establishing the cause-effect relationships for major soils and ecosystems. Assessment of soil quality and resilience must be studied in relation to crop productivity, reduction of soil erosion, control of non-point source pollution, C storage, and emissions of greenhouse gases. Long-term studies that provide data on degradation-induced changes in soil properties are needed for principal soils and ecosystems to contrast degradation vs. resilience relationships and elucidate possible hysteresis in degradation and recovery curves over time.
5. ***Management systems.*** There is also a needed to further identify the best management practices for increasing soil quality and resilience and restoring degraded soils across a wide range of ecosystems. Development of methodologies for retarding soil degradation and accelerating soil resilience is needed for specific disturbances.
6. ***Threshold levels.*** Establishing threshold levels of soil degradation and quality is important. Some soils may not recover beyond certain critical limits of degradation. There is a need to establish functional relationships between crop production and soil erodibility for different management systems and ecoregions. Discussions about soil resilience have limited value unless the threshold levels of soil degradation are well established for each soil type.
7. ***Interdisciplinary studies.*** Soils are part of an ecosystem comprised of animals, plants, and soils. An understanding of the ecological, social, and economical

approaches of soil quality and resilience in relation to agricultural sustainability and environmental quality is fundamental. Soil quality and resilience are interrelated to precision agriculture, biogeochemistry, soil ecology, microbiology, forestry, plant science, and others. Thus, research on soil quality and resilience must be addressed from different perspectives and uses.

8. ***Global climate change.*** Impacts of projected global climate change on soil quality and resilience must be measured and modeled. Development of methods to understand climate change vs. soil quality and resilience relationships is essential to identifying proper management strategies and assessing effects.

21.13 No-Till Farming

Important research and development priorities include the following:

1. Developing a soil suitability guide to no-till farming.
2. Assessing the feasibility of no-till and other conservation technologies adaptable to all ecosystem conditions.
3. Considering social, economic, and cultural issues affecting adoption of no-till farming.
4. Assessing both short- and long-term implications of no-till farming. Positive and negative impacts must be balanced with the goal of conserving and sustaining soil and water resources.
5. Strengthening the local long-term database on no-till performance in regards to crop yields, soil attributes, pest invasions, and environmental quality different management scenarios (e.g., cropping systems).
6. Expanding no-till technology to small-scale farmers in consideration of the differences in technical and financial capacity between large-scale and small scale farmers.
7. Adopting no-till and companion conservation technologies under on-farm conditions for small landholders across contrasting agro-ecological and socio-economic conditions.
8. Designing no-till equipment that fits local conditions and provides uniform results.
9. Assessing the social and economic benefits of no-till farming at all levels of society.
10. Generating more data on the impacts of no-till on C sequestration, greenhouse gas emissions (e.g., CO_2, N_2O, and CH_4), soil productivity, and possible impacts on the projected global climate change.

21.14 Soil Organic Carbon

Research in soil conservation and management in relation to C storage and dynamics would benefit from the following approaches:

1. Establishing relationships of soil erosion by water, wind, and tillage with soil organic C dynamics and storage under different soils, ecosystems, and climate conditions.
2. Determining fate of eroded C under different scenarios of tillage and cropping systems.
3. Understanding distribution of C along different landscape positions to quantify the magnitude of tillage-induced changes in C dynamics and storage.
4. Modeling the fate of C transported by erosion to determine whether erosion is source or sink of C.
5. Collating experimental data on C dynamics within the depositional areas in eroded landscapes to elucidate the magnitude of C accumulation and its significance to total C pool.
6. Assessing C pool and fluxes for a wide range of soils under varying topographic and climatic conditions. This approach could also enhance the scientific understanding of the dependence of C sequestration on soil intrinsic characteristics (e.g., soil texture, drainage, topography) and clarify findings that show that rate of C sequestration may be either low or negligible by conversion of plow tillage to no-till in clayey and even in some silt loam soils
7. Collating information on C sequestration in no-till farming under farmer's fields rather than only under small research plots. Comparatively less is known about the no-till farming implications on C sequestration under on-farm conditions. This information important to discern the potential of no-till in sequestering C at a large scale.
8. Elucidating the views that higher C under no-till observed in surface layers might due to shallow sampling protocols by assessing no-till impacts on C sequestration for the entire soil profile rather than only for the shallow surface soil ($<$30 cm depth).
9. Strengthening database in no-till potential across a regional scale especially now when large areas of croplands are being converted to long-term no-till systems based on the premise, in part, that no-till sequesters C and may open economic opportunities of trading C for farmers through the Chicago Climate Exchange (CCX). Such a database on C dynamics is also needed as baseline information for modeling C sequestration in croplands at regional and national scales.
10. Obtaining information on the performance of bioenergy crops under marginal or reclaimed lands for reducing CO_2 emissions and sequestering C.
11. Developing new techniques of *in situ* C measurement such as LIBS, IRS, INS and refining others by conducting site-specific calibration across a wide range of soils to make them robust tools for C assessment.
12. Assessing residence time, recalcitrance, and deep burial of C in agricultural soils.
13. Evaluating the importance of biochar to sequestering and mitigating global climate change.

21.15 Deforestation

Management protocols which must be reinforced to halt tropical deforestation include the following:

1. Accelerate reforestation and afforestation of degraded forestlands and stimulate regrowth of native vegetation by managing secondary forests and adding diverse plant species. While short-rotation woody plantations provide goods, their benefits lag behind a mixture of native forest species of higher diversity which not only provide more wood and non-wood products but also are more beneficial to wildlife habitat and environmental quality protection.
2. Create protected areas of forests to conserve diversity of flora and fauna. Model forests must be established to show the best forest management practices across representative regions.
3. Increase productivity of current agricultural lands to reduce shifting cultivation, and thereby further deforestation.
4. Develop a data-base with information and guidelines pertinent to the extent of deforestation, control measures, management of tree species (e.g., selection, planting) and economic returns. Long-term economic returns from high-quality wood products from diversified natural forests can be higher than those from fast-growing trees.
5. Identify appropriate forest management scenarios and conservation strategies for a wide range of soils and climate conditions.
6. Establish agroforestry practices integrated with agricultural crops or livestock as an alternative to pure forest systems. Agroforestry systems with diversified tree species in conjunction with shade-tolerant annual crops provide diversified economic returns.
7. Monitor existing forest resources to determine rates of deforestation and restoration.
8. Use improved technologies such as remote sensing, satellital images, geographic positioning systems, process-based models, and geographic information systems to assess forest cover and degradation.
9. Develop standards, principles, and guidelines for sustainable management of forestlands.
10. Identify strategies for moving from site-specific reforestation to landscape levels of reforestation. Small and site-based reforestation practices are laudable initiatives but reforestation at larger scale or landscape level is needed to effectively reverse soil degradation, conserve biodiversity, and improve farm economy and the environment.
11. Validate best management practices in cleared forestlands such as reduced tillage, no-till, contour plantings, composting, cover crops, and deferred grazing.
12. Establish windbreaks, hedgerows, and riparian buffers as an integral part of forest management. Buffers are essential to reduce streambank erosion. Stream channels and sensitive areas must be fenced to reduce sediment load.

13. Establish vegetative cover and mechanical control practices to minimize erosion from road banks in forestlands. Roads must be constructed on ridges or on gentle slope gradients with the least amount of soil disturbance and with adequate runoff and soil erosion control measures. Mulching of disturbed soil, during vegetation establishment, is an option to stabilize roadbanks.
14. Strengthen and develop extension programs to extend practices of reforestation, afforestation, and other forest management strategies to reduce the need for additional land clearing. Given the large scale of deforestation, a broad participation of landowners and users must be undertaken for an effective establishment of programs of restoration.
15. Assess the impact of native tree and shrub species to providing diverse goods and services (e.g., fruit, bioenergy, fiber) to reduce pressure on primary forests. Large-scale plantations of trees are the best option to reverse deforestation.
16. Link afforestation with improvement of environment and rural economy. Develop policies that provide incentives to farmers involved in reforestation, which provide ecosystem services (e.g., C sequestration, biodiversity conservation, and soil and water conservation) to the whole society.
17. Restore degraded, abandoned, marginal, and deforested lands with forest farms. Extent of degraded and abandoned has increased in recent years, which provides more land available for large-scale forest plantations aimed at reducing poverty in rural areas.
18. Introduce proper logging regulations, regulate access to conservation areas, and improve recycling of wood processed by-products.

21.16 Abrupt Climate Change

Further understanding of climate change impacts on soil erosion and soil and water conservation practices is required as follows:

- Conducting field experiments to simulating effects of various scenarios of climate change on runoff, soil erosion, soil erodibility, soil organic C dynamics, and crop production factors across a wide range of latitudes and contrasting scenarios of management and soil type to gain a better understanding of implications of global warming.
- Developing models to assess the impacts of the projected climate change by integration of climate models from different disciplines including geology, climatology, agronomy, and environmental sciences.
- Modeling of critical times and probability of occurrence of intense rain storms, soil vulnerability to erosion, and threshold levels of damage caused by soil erosion across representative soil types, management practices, and cropping systems. Most modelers have studied scenarios of climate change on soil erosion by controlling only a few erosion factors, specifically rainfall, temperature, and CO_2 emissions and have not simulated interactive processes including soil response,

adaptations/resistance/resilience of plants, and possible genetic changes and photosynthetic characteristics of plants under the new climate.

- Developing models to account for climate change-induced modifications in soil processes and properties affecting soil erodibility.
- Improving climate models by including simulations of the expected farmer's adaptations (e.g., earlier planting dates, longer cropping seasons) to new climate.
- Simulating future soil erosion by accounting the complex soil-plant-atmosphere relationships. Simulations of erosion rates must account for shifts in tillage systems and cropping systems, changes in soil properties, and incidence of pests and diseases as affected by climate change. Current modeling approaches of climate change impacts on soil erosion are too simplistic to account for the complexity of agricultural systems and capture the temporal resolution of soil erosion predictions.

21.17 Modeling

This is the age of modeling in all aspects of research. Modeling is an essential component of soil and water conservation research and practice. Use of models is expected to continue to increase as a tool to predict implications of different management scenarios on soil and water conservation and global climate change. For example, models are not only used for predicting runoff and soil erosion from croplands but also from other land use (e.g., grasslands, forestlands, minesoils) systems.

Whilst models have advanced our understanding of the erosion processes, current approaches are mostly empirical and available models have numerous limitations for capturing the complexity of natural systems across large areas. So far, empirical models of soil erosion are more widely used than process-based models. This is because of the large database required by process-based models, which are not always available for all ecosystems and regions. Most of the existing models used in soil and water conservation were designed for research purposes and not for solving specific problems in large-scale ecosystems. The USLE is the most widely used erosion model, but it uses empirical approaches. Despite the extensive work that went on collecting data from $>$10,000 plots for developing, the USLE and RUSLE often under- or over-estimate soil erosion rates, and their results can not be extrapolated to larger scale. The simplicity and accessibility of these empirical models sacrifices the refinement of the model for large-scale systems.

Modeling of soil and water conservation can be enhanced by:

1. Developing and improving comprehensive and physically-based models to scientifically estimate the magnitude of soil erosion in space and time.
2. Expanding the database on input parameters for erosion modeling. Modern soil surveys and ample data on crop production, conservation practices, tillage management and cropping systems, and other parameters are needed.

3. Creating management-specific models to increase their applicability and use across local and regional scales.
4. Expanding the use of remote sensing and GIS as forefront technological tools to monitor, evaluate, and implement conservation practices.
5. Testing and validating the performance of models in association with GIS and other models.
6. Refining field and watershed models to provide better estimations of runoff and sediment yield and redistribution.
7. Modeling runoff and soil erosion at the landscape field and watershed scales.
8. Simulating the dynamic nature of land use and management systems and their response to soil and water conservation technologies.

21.18 Soil Management Techniques for Small Land Holders in Resource-Poor Regions

Restoration of degraded soils and management of existing productive soils are high priorities particularly for small-land holders, which farm an average of 2 ha. Food insecurity from the decline in crop yields due to nutrient mining and nutrient depletion by soil erosion is a pressing concern in poor regions of the world such as is the case in Africa. Since expansion of cultivated area is not possible in most regions due to the scarcity of prime agricultural soils, food security must be improved by increasing the crop production per unit area. Principal challenges are to reduce soil erosion to permissible levels and to improve fertility of soils managed by resource-poor farmers. Techniques of soil management for small farmers vary depending upon the ecosystem, topography, and climate. Various measures can be used for managing soil erosion and improving soil fertility. Among these are (Mafongoya et al., 2006):

1. ***Terraces.*** These structures intercept, retain, slow, and divert runoff to safe outlets, reducing soil erosion and loss of nutrients. There are different types of terraces that can be used by small farmers including conservation bench terraces, hillside ditches, orchard terraces, and intermittent and continuous terraces.
2. ***Rainfed ponds.*** Rainwater harvesting is a strategy to reduce soil erosion, store water for crops, and increase crop yields in sloping fields. During rainy seasons, runoff and rainwater can be harvested by constructing ponds. The stored water is used for irrigation and growing crops in dry seasons.
3. ***Inorganic fertilizers.*** Application of inorganic fertilizer at recommended rates and proper times increase efficiency of use while increasing crop yields. Efficient use and management of inorganic fertilizers can reduce environmental degradation. Costs of fertilizers must be reduced to allow accessibility and use. Applying 9 kg ha^{-1} of N can increase crop yields by as much as 50% if it is applied at the right time and place (ICRISAT, 2006). The effectiveness of fertilizers is a function of rainfall amount, type of crops, type of soil and tillage management.

4. ***Animal manure.*** Manuring is one of the oldest techniques to improve soil fertility. Quality and quantity of manure are often low in developing countries, depending on the type of animal and forage quality. The efficiency of the low quantity of animal manure produced can be improved by placing manure directly into the holes or furrows where plants will be grown as an alternative to broadcasting all over the field. As much as 60% of N and 10% of P can be lost during broadcasting and lack of proper management of manure (Mafongoya et al., 2006). This approach not only improves the efficiency of manure use but also reduces losses of manure-derived products through erosion, volatilization, and leaching. Manure management guidelines must be developed to reduce close contact of seeds with manure and rates of application.
5. ***Grain legumes.*** Incorporating grain legumes into traditional cropping systems is vital to improve soil fertility and N cycling. Grass and tree legumes when rotated or intercropped with row crops can be used as green manures. The biological nitrogen fixation by legumes contributes to soil with N and can reduce needs for using inorganic fertilizers. Growing legumes is an ecological and cost-effective strategy to restore soil fertility. Economical benefits of rotating row crops with grasses or tree legumes must be determined as well as the guidelines of establishment and management developed.
6. ***Agroforestry practices***. Planting trees around and within croplands reduces soil water and wind erosion. Trees can also store N in soil through biological nitrogen fixation. Large-scale adoption of fertilizer trees is a potential solution to replenish N to nutrient-starved soils. *Sesbania, Tephrosia, Gliricidia, Leucaena, Calliandra, Senna,* and *Flemingia* are some of the agroforestry species used for improving soil fertility in Africa ((Mafongoya et al., 2006). More aggressive expansion of agroforestry technology is needed as companion to grain legumes.
7. ***Integrated nutrient management practices***. Combining inorganic fertilizers and organic amendments is a better alternative to the use of either inorganic fertilizers or organic amendments (e.g., manure, compost) alone. The interaction of both nutrient sources reduces excessive use of inorganic fertilizers and improves nutrient-supplying capacity of organic amendments. An integrated approach of nutrient management involves the methods of application, timing, amount, and type of fertilizers in combination with grain legumes and agroforestry practices.
8. ***Tillage management***. No-till, reduced or minimum, mulch, and strip tillage are recommended tillage systems to restore degraded soils.
9. ***Residue management***. Residue return following harvest is important to maintain a protective cover and reduce wind and water erosion. Residue cover is insufficient in poor regions due to limited residue production and competing uses for residue. Returning crop residues and planting grass and legume trees can increase the amount of residue cover.
10. ***Conservation buffers***. Filter strips, grass barriers, riparian buffers, windbarriers, and field borders protect soil from erosion. Integration of grass barriers with food crops paralleling rows of crops reduce removal of sediment while buffers established at the lower end of fields reduced off-site transport of sediment.

11. ***Cropping systems***. Crop rotations, multi-cropping, strip cropping, contour farming, contour strip cropping, and cover cropping are strategies to conserve and water. Dense canopy and high-biomass producing crops intercept raindrops and protect the soil surface. Systematic arrangement of crops in strips, on the contour, and growing various but different crops per year reduce soil erosion.

Summary

Despite the significant progress achieved in understanding soil erosion processes and factors as well as development of erosion control practices, rates of soil erosion are still above the permissible levels and thus warrant further attention. Water pollution with sediment and chemicals is an ongoing concern particularly in industrialized regions. Technologies must be improved or engineered to reduce the current levels of pollutant delivery to downstream water resources. Food insecurity is real in resource-poor regions of the world where farmers have no choice to cultivate meager, eroded, and infertile soils for obtaining their daily food.

A multidisciplinary approach across the diverse political, social, economic, and biological conditions of each region is needed to design erosion control and management strategies to achieve an effective soil and water conservation. The declining funding support for research on soil erosion must be reversed to develop new technologies of soil erosion control. Economical soil conservation practices that are accessible to farmers must be developed, refined, and promoted including crop residue management, biological practices, use of buffer strips, agroforestry, organic farming, and others.

Soils must be conserved and managed to provide food, produce biomass for biofuel, sequester soil organic C for reducing the atmospheric build-up of greenhouse gases, absorb air and water pollutants to maintain environmental quality, and improve wildlife and biodiversity. Conditions will change and new challenges will arise, but the hope is that science and technology will deliver more refined and proper measures for controlling and reversing soil degradation. Land stewardship and improved technologies are the basic strategies to conserve soil and water for the present and future generations. Managing soil erosion is imperative and there is no time to be complacent about the dangers that erosion poses to humanity. Only a decisive participation of farmers, soil conservationists, policymakers, and the whole society can reverse the downward spiral of soil erosion and its consequences.

Study Questions

1. Describe some major soil erosion-induced environmental concerns in the world.
2. Discuss the strategies to counteract the environmental concerns in Prob. 1.
3. What are the differences between erosion control practices and soil conservation practices.?

4. Discuss how soil erosion can be effectively controlled when demands for food, fiber, and energy production are raising rapidly.
5. Discuss the advantages of disadvantages of innovative conservation practices such as no-till and organic farming for reducing water pollution and the projected global climate change.
6. Cite the specific strategies for improving performance of current conservation practices (e.g., amendments, grass buffers, agroforestry, soil conditioners).
7. Discuss the importance of modeling soil erosion.
8. Discuss the possible impacts of the global climate change on soil erosion.
9. What are the projected impacts of producing biomass for biofuel on a large scale on soil and water conservation.?
10. Explain the potential of conservation tillage and cropping systems for sequestering C in soil to mitigate the atmospheric concentration of greenhouse gases.

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Appendix A

Abbreviations of Some of the Words Frequently Use in the Textbook

A	Average annual soil loss
AGNPS	GNPS Agricultural Non-Point Source Pollution Model
AnnANPSPL	Annualized Agricultural Non-Point Source Pollutant Loading
ANSWERS	Areal Nonpoint Source Watershed Environment Response Simulation
BNF	Biological N fixation
C	Carbon
C-factor	Cover and management factor
CEC	Cation exchange capacity
CCX	Chicago Climate Exchange
CI	Cone index
CN	Runoff curve number
CROPSYST	Cropping Systems Simulation Model
CRP	Conservation Reserve Program
EC	Electrical conductivity
EGEM	Ephemeral Gully Erosion Model
EPIC	Erosion Productivity Impact Calculator
EUROSEM	European Soil Erosion Model
GHG	Greenhouse gas
GIS	Geographic information systems
GLEAMS	Groundwater Loading Effects of Agricultural Management Systems
GUEST	Griffith University Erosion System Template
ICRAF	The International Center for Research in Agroforestry
INS	Inelastic Neutron Scattering
IRS	Infrared Reflectance Spectroscopy
K	Erodibility factor
K_{sat}	Saturated hydraulic conductivity

LER	Land equivalent ratio
LIBS	Laser Induced Breakdown Spectroscopy (LIBS)
LISEM	Limburg Soil Erosion Model
LS	Topographic factor
Mha	Million hectares
MUSLE	Modified Universal Soil Loss Equation
MWD	Mean weight diameter
NAC	National Agroforestry Center
NT	No-till
P	Phosphorus
P	Support practices
PAM	Polyacrylamide
PCA	Principal component analysis
PI	Productivity index
PLS	Partial least-squares analysis
PTFs	Pedotransfer functions
R	Rainfall and runoff erosivity index
RUSLE	Revised Universal Soil Loss Equation
RWEQ	Revised WEQ
S	slope
SAR	Sodium absorption ratio
SAT	saturation factor
SOC	Soil organic carbon
SQI	Soil quality index
SWAT	Soil and Water Assessment Tool
TSD	Topsoil depth
USDA-NRCS	United States Department of Agriculture-Natural Resources Conservation Service
USLE	Universal Soil Loss Equation
WATEM	Water and Tillage Erosion Model
WEPP	Water Erosion Prediction Project
WEPS	Wind Erosion Prediction System
WEQ	Wind Erosion Equation
WSA	Water-stable aggregates

Appendix B

Common and Scientific Names of Plants Used in the Textbook

Acacia	*Acacia spp.*
African marigold	*Tagetes erecta*
Aleppo pine	*Pinus halepensis*
Alfalfa	*Medicago sativa*
Amaranth	*Amaranthus spp.*
Bahiagrass	*Paspalum notatum*
Banana	*Musa paradisiaca*
Barley	*Hordeum vulgare*
Beach sheoak	*Casuarina equisetifolia*
Beans	*Phaseolus vulgaris*
Bermuda grass	*Cynodon dactylon*
Big bluestem	*Andropogon gerardi*
Birdsfoot trefoil	*Lotus corniculatus*
Black walnut	*Juglans nigra* L.
Blackberry	*Rubus ursinus*
Blue grama	*Bouteloua gracilis*
Blueberry	*Vaccinium* spp.
Bluegrass spp.	*Poa spp.*
Buffalo grass	*Bouteloua dactyloides*
Cajanus	*Cajanus Adans*
Calapo	*Calopogonium mucunoides*
Calliandra	*Calliandra calothyrsus*
Canada bluegrass	*Poa compressa*
Canola	*Brassica napus*
Carpet grass	*Axonopus compressus*
Carrots	*Daucus carota*
Cassava	*Manihot esculenta*
Centipedegrass	*Eremochloa ophiuroides*
Centro	*Centrosema pubescens*
Chickpea	*Cicer arietinum*
Chicory root	*Cichorium intybus*

Clover	*Trifolium repens*
Cocklebur	*Xanthium strumarium*
Coconut	*Cocos nucifera*
Coffee	*Coffea L.*
Corn	*Zea mays*
Cotton	*Gossypium* spp.
Cottonwood	*Populus deltoids*
Cowpea	*Vigna unguiculata*
Creeping grass	*Cynodon plectostachyum*
Crimson clover	*Trifolium incarnatum*
Dames rocket	*Hesperis matronalis*
Downy brome	*Bromus tectorum*
Dry bean	*Phaseolus vulgaris*
Eastern gamagrass	*Tripsacum dactyloides*
Elephant grass	*Pennisetumpolystachion*
Eucalypts	*Eucalypts spp.*
Evening primrose	*Oenothera macrocarpa*
Fababean	*Vicia faba*
Field pea	*Pisum sativum*
Field thistle	*Cirsium discolor*
Flemingia	*Flemingia congesta*
Garlic	*Allium sativum*
Giant cane	*Arundinaria gigantea*
Ginseng	*Panax L.*
Gliricidia	*Gliricidia sepium,*
Gmelina	*Gmelina arborea*
Goldenrod	*Solidago*
Gray sheoak	*Casuarina glauca*
Guinea grass	*Panicum maximum*
Hairy vetch	*Vicia villosa*
Horseweed	*Conyza canadensis*
Huckleberry	*Gaylussacia* spp.
Indian grass	*Sorghastrum nutans,*
Indian oak	*Tectona grandis*
Italian ryegrass	*Lolium multiflorum*
Kallar grass	*Leptochloa fusca*
Kentucky bluegrass	*Poa pratensis*
Kinghead ambrosia	*Ambrosia trifida*
Kudzu	*Pueraria phaseoloides*
Leek	*Allium porrum*
Lentils	*Lens culinaris*
Lespedeza	*Lespedeza cuneata*
Lettuce	*Lactuca sativa*
Leucaena	*Leucaena leucocephala*
Mesquite	*Prosopis juliflora*

Millet	*Pennisetum americanum*
Mimosa	*Albizia julibrissin*
Miscanthus	*Miscanthus sinensis*
Monterey pine	*Pinus radiata*
Mountain immortelle	*Erythrina poeppigiana*
Mucuna	*Stilozobiun niveun*
Natural fallow	*Chromolaena odorata*
New England aster	*Aster novae-angliae*
Oat	*Avena sativa*
Onion	*Allium cepa*
Orchard grass	*Dactylis glomerata*
Pea	*Pisum sativum*
Peanuts	*Arachis hypogaea*
Pheasantwood	*Senna siamea*
Pigeon pea	*Cajanus cajan*
Pine	*Pinus* spp.
Pineapple	*Ananas comosus*
Poplar	*Populus* spp.
Potato	*Solanum tuberosum*
Red clover	*Trifolium pratense*
Red fescue	*Festuca rubra*
Red gum	*Eucalyptus tereticornis*
Rice	*Oryza sativa*
Rryegrass	*Lolium multiflorum*
Rye	*Secale cereale*
Senna	*Senna siamea*
Sesbania	*Sesbania bispinosa*
shiitake mushrooms	*Lentinula edodes*
Smooth bromegrass	*Bromus inermis*
Sorghum	*Sorghum bicolor*
Soybean	*Glycine max*
Squash	*Cucurbita* spp.
Style	*Stylosanthes gracilis*
Sudangrass	*Sorghum bicolor*
Sugar beet	*Beta vulgaris*
Sugarcane	*Saccharum officinarum*
Sunflower	*Helianthus annuus*
Sunnhep	*Crotalaria juncea*
Surgarbeet	*Beta vulgaris*
Swamp oak	*Casuarina obesa*
Swamp white oak	*Quercus bicolor*
Sweet potatoes	*Ipomoea batatas*
Sweetclover	Melilotus officinalis
Switchgrass	*Panicum virgatum*
Tall fescue	*Festuca arundinacea*

Taro	*Colocasia esculenta*
Tephrosia	*Tephrosia vogelii*
Timber tree	*Terminalia amazonia*
Timothy	*Phleum pretense*
Tomato	*Solanum lycopersicum*
Tropical kudzu	*Pueraria phaseoloides*.
Velvet bean	*Mucuna pruriens*
Vetch	*Coronilla* and *Vicia* spp.
Vetiver grass	*Vetiveria zizanioides*
Watermelon	*Citrullus lanatus*
Wheat	*Triticum aestivum*
White clover	*Trifolium repens*
White sweetclover	*Melilotus alba*
Wild kulthi	*Atylosia scarabaeoides*
Willow	*Salix* spp.
Yam	*Dioscorea* spp.

Color Plates

Plate 1 Soil erosion not only reduces soil fertility, crop production, and biodiversity but also alters water quality and increases risks of global climate change and food insecurity (Courtesy USDA-NRCS)

Plate 2 Wind erosion reduces vegetative cover and forms large sand dunes in arid regions (Photo by H. Blanco)

Plate 3 Runoff sediment pollutes nearby water sources (Courtesy USDA-NRCS)

Plate 4 Air pollution during the Dust Bowl (Courtesy USDA-NRCS)

Plate 5 Map of Africa showing areas (*dark*) where soil degradation is a serious problem and population exceeds the land's carrying capacity (After Holden, 2006)

Plate 6 A cropland affected by rill and interrill erosion (Courtesy USDA-NRCS)

Plate 7 Concentrated runoff forms gullies (Courtesy USDA-NRCS). Channels without hydraulic roughness elements erode at faster rates with incoming runoff than those nested with deep plant roots and rocks. Gullies are expanded by steep water fall at the gully heads, called headcut, and by gradual lateral erosion and sloughing of the gully sides

Plate 8 Corn field severely affected by streambank erosion (Courtesy USDA-NRCS). Saturated soils along streambanks slump readily under concentrated runoff, which causes scouring and undercutting of streambanks and expansion of water courses

Plate 9 The Swanson type rotating boom rainfall simulator (Photo by H. Blanco). The simulator booms are equipped with nozzles positioned at radii of 1.5, 3.0, 4.5, 6.0, and 7.6 m. Booms and nozzles rotate in a circle, and the wetted diameter is about 16 m

Plate 10 Wind erosion creates sand dunes in arid regions (Photo by H. Blanco)

Plate 11 Well-designed windbreaks in North Dakota reduce wind erosion (Courtesy USDA-NRCS)

Plate 12 Three-row windbreaks are effective soil erosion control measures (Photo by H. Blanco)

Plate 13 Plowing shifts soil downhill (Courtesy T.E. Schumacher, South Dakota State Univ.)

Plate 14 Tillage erosion increases with tillage intensity and slope gradient (Courtesy T.E. Schumacher, South Dakota State Univ.)

Plate 15 Rye as a cover crop for corn-soybean rotation in Pennsylvania (Photo by H. Blanco)

Plate 16 Crop residues protect soil from cracking, crusting, and surface sealing (Photo by H. Blanco)

Plate 17 Corn produces large amounts of residues (Photo by H. Blanco)

Plate 18 Spraying animal manure slurry is common for improving the soil fertility (Courtesy USDA-NRCS). Manure application at optimum rates is an important to reducing risks of water pollution

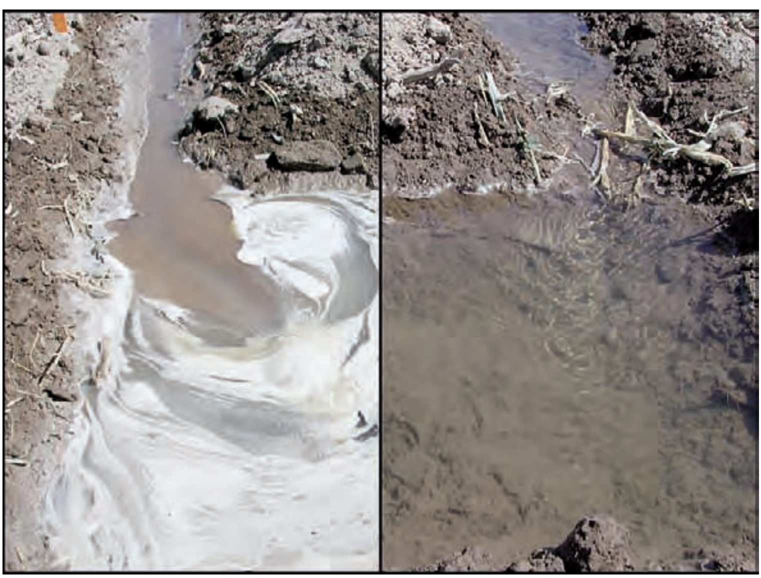

Plate 19 Use of PAM in irrigation water reduces runoff sediment and soil erosion (Courtesy of the USDA-ARS, Northwest Irrigation and Soils Research Laboratory, Kimberly, ID)

Plate 20 Corn-alfalfa rotation to conserve soil and improve soil fertility in central Ohio (Photo by H. Blanco)

Plate 21 Row crops involving onion (*left*) and corn (*right*) with little or no residue cover (Photo by H. Blanco). The bare interrows with wide spacing can develop rills

Plate 22 Contour farming reduces erosion and improves soil productivity in sloping fields (Courtesy USDA-NRCS)

Plate 23 Contour stripcropping protects the soil from erosion and improves land aesthetics (Courtesy USDA-NRCS)

Plate 24 Comparison between a plow tillage (*left*) and no-till (*right*) system on a silt loam (Photo by H. Blanco)

Plate 25 Long-term no-till soil (*left*) next to an intensively moldboard plowed soil (*right*) in a clay loam soil (Photo by H. Blanco)

Plate 26 Cool soils under heavy residue mulch slow germination and emergence of corn in no-till systems (Photo by H. Blanco). Soil temperature dynamics in no-till soils under different climatic conditions and seasons must be understood to properly manage crop residues for conserving soil and water. Wet, cool, and clayey soils are the most adversely affected by heavy residue mulching

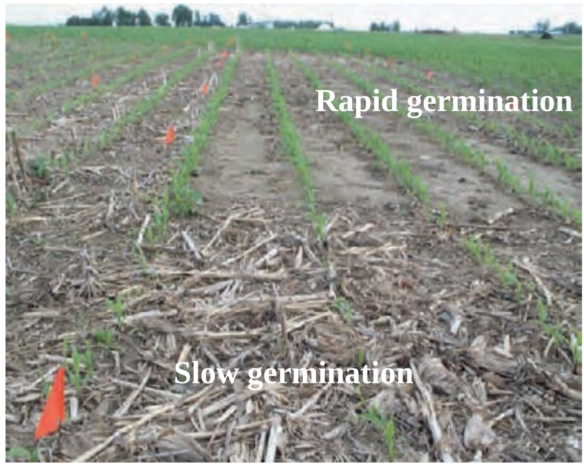

Plate 27 A ridge tillage field used for soil and water conservation (Courtesy USDA-NRCS). Each ridge supports one single row of plants. Tillage and planting are done on the same ridges year to year

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Big Bluestem **Switchgrass** **Indian grass**

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Plate 59 A lake severely affected by algae blooms (Courtesy USDA-NRCS)

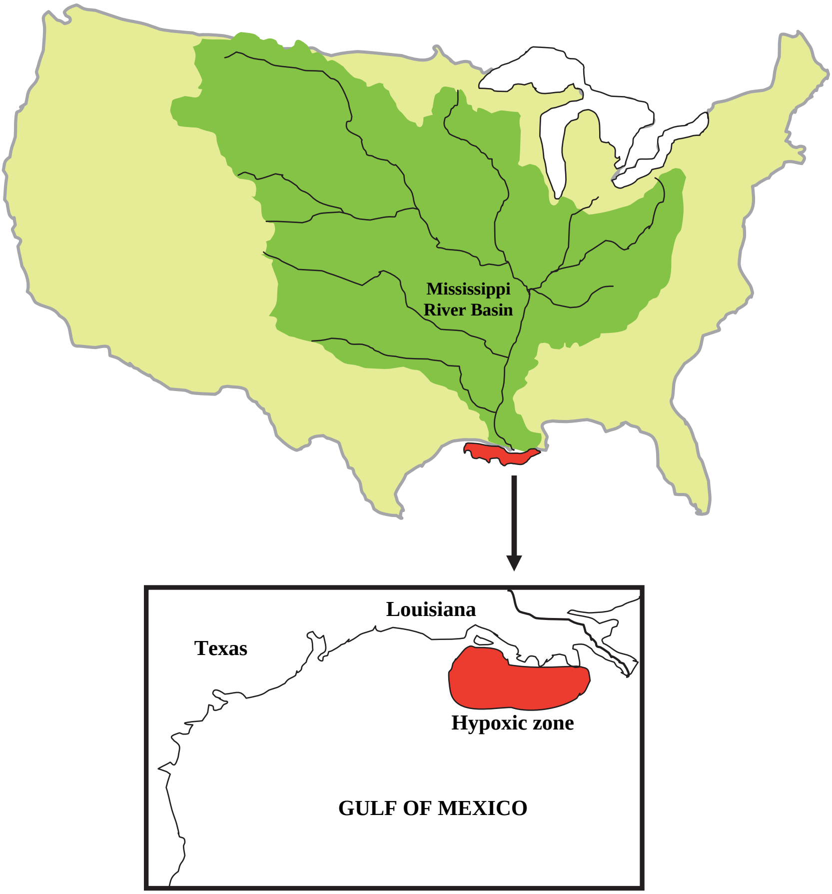

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Plate 67 The closed chamber method consists of a gas sampling chamber made of PVC with a bottom section (30 cm long × 15 cm diameter) inserted into the ground, and a lid equipped with a gas sampling port (Photo by H. Blanco). Air samples withdrawn from the chamber are stored in evacuated vials for the soil gas (e.g., CO_2, CH_4) analyses

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