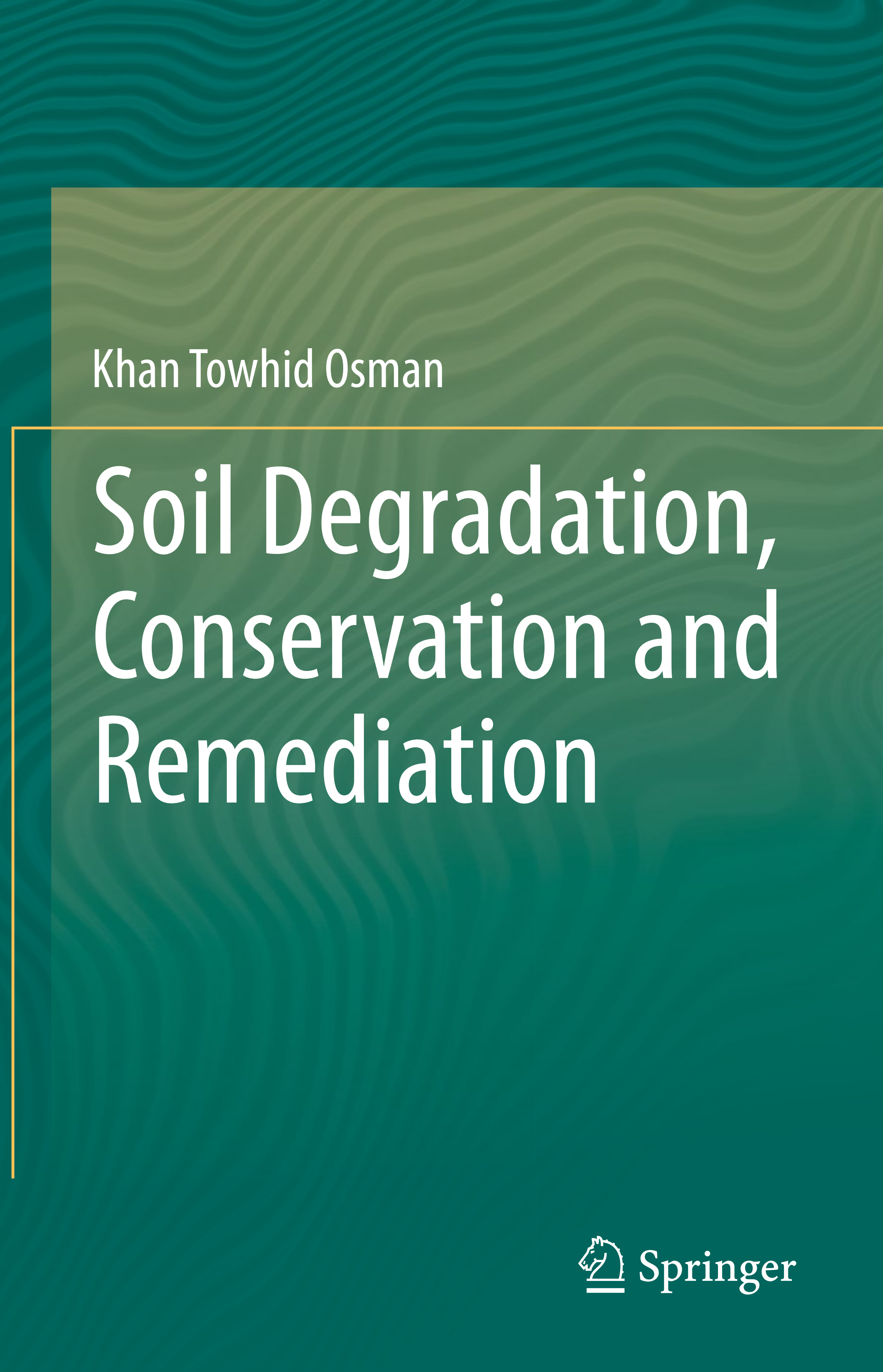

Khan Towhid Osman

Soil Degradation, Conservation and Remediation

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Md. Bazlul Karim Khan and Rowshan Ara Begum
Who would have been the proudest of their son's work

Preface

Recently, I wrote a book entitled *Soils: Principles, Properties and Management* which Springer published in December 2012. The original manuscript had a long chapter of more than 75 normally typed pages on soil resources and degradation. While writing the chapter, I collected more than adequate literatures on the levels and impacts of soil degradation worldwide. I figured it would be a good idea to extend it with necessary details, examples, tables, and figures into a full-fledged book on soil degradation, conservation, and remediation should an opportunity ever present itself. I did so eventually, and while writing, I endeavored to give lucid accounts of the principal processes of soil degradation, its effects on soil quality, plant production and human health, and methods of improvement of degraded soils.

The decline in soil quality due to human interventions in soil, water, and environment is called human-induced soil degradation. H. Eswaran et al.'s observation on the looming significance of soil degradation studies is noteworthy: "Soil degradation has been a major global issue during the 20th century and will remain high on the international agenda in the 21st century. The importance of soil degradation among global issues is enhanced because of its impact on world food security and quality of the environment" (http://soils.usda.gov/use/worldsoils/papers/land-degradation-overview.html). Deforestation of fragile lands, overexploitation of vegetation and biomass resources, shifting cultivation, overgrazing, unbalanced fertilizer use, non-adoption of soil conservation management practices, use of ground water in excess of the capacity for recharge, inappropriate irrigation system, and indiscriminate disposal of wastes are some of the factors responsible for soil degradation. Physical deterioration of soil including surface sealing, hardsetting and compaction, water and wind erosion, and chemical soil degradation including nutrient depletion, acidification, salinization, and soil pollution are the chief processes of soil degradation. Decline in soil productivity and fall in crop quality are the measurable impacts of soil degradation. The productivity of some lands has declined by as much as 50 % due to soil erosion and desertification. Mean yield reduction due to erosion in Africa is 8.2 %. Annual loss in productivity due to water erosion is estimated at 36 million tons of cereal equivalent to US$5,400 million and due to wind erosion to US$1,800 million in South Asia. It is estimated that the total annual cost of erosion from agriculture in the USA is about US$44 billion per year. On a global scale, the annual loss of 75 billion tons of soil costs the world about US$400 billion per year.

Soil Degradation, Conservation, and Remediation is intended for undergraduate and graduate students of Soil Science, Agricultural Sciences, Forestry, Ecology, Geography, and Environmental Sciences. The processes and impacts of soil degradation have been dealt with in this book in sufficient details. Chapter 1 describes global soil resources, land capability and soil quality classes, soil orders, global arable land, causes and types of soil degradation, and laws of sustainable management of soil. Chapter 2 deals with factors and processes of physical deterioration of soil, including surface sealing, surface crusting, hardsetting, compaction, effects of crusting and compaction, and decrusting and decompaction of soil. Chapter 3 narrates the causes, factors, and processes of water erosion. Methods of soil conservation including amendments, conservation farming, cover crops, no tillage, minimum tillage, mulching, contour cropping, strip cropping, contour-strip cropping, SALT, terracing, and grassed waterways have been explored with examples and data. In Chap. 4, causes, effects, and processes of wind erosion are described. Measures to control wind erosion and dune stabilization are shown with a good number of illustrations. Chapter 5 emphasizes chemical degradation of soil including nutrient depletion, acidification, and salinization (but not soil pollution which is narrated in detail in Chap. 6).

My colleagues Dr. Abul Kashem, Mr. Jajar Afsar, and Md. Enamul Haque of the Department of Soil Science, University of Chittagong, have indebted me with inspiration and useful suggestions on the manuscript. Thanks to them are due.

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Contents

Abbreviations

AMD	Acid Mine Drainage
ASSOD	Assessment of human-induced soil degradation for South and Southeast Asia
BTEX	Benzene, toluene, ethylbenzene, xylene
DCE	Dichloroethylene
DCPA	Dichloro propanilic acid
DDT	Dichlorodiphenyltrichloroethane
EDC	Endocrine disrupting chemicals
FAO	Food and Agriculture Organization of the United Nations
GLADA	Global assessment of land degradation in arid lands
GLASOD	Global assessment of human-induced soil degradation
ISRIC	International Soil Reference and Information Centre
LADA	Land Degradation Assessment for Dryland Areas
NPE	Nonylphenol ethoxylates
NRCS	Nature Resource Conservation Service
PAE	Phthalate esters
PAHs	Polycyclic aromatic hydrocarbons
PCB	Polychlorinated biphenyl
PCDD	Polychlorinated dibenzodioxin
PCDF	Polychlorinated dibenzofurans
PCE	Polychloroethylene
PCN	Polychlorinated naphthalenes
POP	Persistent organic pollutant
RUSLE	Revised Universal Soil Loss Equation
SALT	Sloping Agricultural Land Technology
SOTER	Soil and Terrain Digital Database
TCDD	Tetrachloro-*p*-dibenzodioxin
TCE	Trichloroethylene
TEPP	Tetraethyl pyrophosphate
UNCCD	United Nations Convention to Combat Desertification
UNEP	United Nations Environmental Program

UNESCO	United Nations Educational, Scientific and Cultural Organization
USA	United States of America
USDA	United States Department of Agriculture
USLE	Universal Soil Loss Equation
VC	Vinyl chloride
WHO	World Health Organization
WRB	World Reference Base

Chapter 1
Soil Resources and Soil Degradation

Including rocky surfaces, deserts, and ice-covered areas, there is 130,575,894 km^2 land area in the globe. About 38.5 million km^2 (29.45 % of the earth's ice-free land surface) is too dry for human habitation, and about 20.2 million km^2 (15.46 %) of the land occurring in the cold tundra zone is not much suitable for normal agriculture. Saline and alkaline soils occur in 3,105,000 km^2 (2.4 % of the land surface), and soil acidity affects 18,420,100 km^2 (14.1 % of the total land). Only about 12 % of the land surface is suitable for food and fiber production, 24 % is grazing land, and about 31 % is forestland; the remaining 33 % has many constraints for most uses. The lands are classified into eight land capability classes based on their productivity, limitations, climate, and soil conservation requirements. Class I land is the most suitable agricultural land, and Classes VII to VIII are not suited for cultivation. Agricultural soils are again classified into nine land quality classes based on the integration of three soil performance classes and three soil resilience classes. Land quality I is the prime land. Suitability of cropping and productivity gradually decrease from Class I to Class IX. The global arable land area is estimated to be 1.351 billion hectares, and 38 % of the arable land has been degraded at variable intensities. The principal cause of soil degradation is the land mismanagement. GLASOD identified five main causes of soil degradation, including deforestation, overgrazing, mismanagement of agricultural land, overexploitation of vegetation, and (bio-) industrial activities. There are five types of land/soil degradation: water erosion, wind erosion, physical deterioration, chemical deterioration, and degradation of biological activity. With this background, ten "laws of sustainable soil management" have been suggested.

1.1 Soil as a Resource

Let us add a few words to Rosewell (1999). To him and to all of us, soil is one of the world's most valuable natural resources. It is essential to all life forms on this planet. It provides a physical matrix, chemical environment, and biological setting for water,

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nutrient, air, and heat exchange for organisms. It provides us with food, fodder, wood, and fiber. Almost 96 % of human food is obtained from the soil (Pimental and Hall 1989). A great number of antibiotics are produced by soil microorganisms. Soil acts as a recycler of materials and as a purifier of water. Soils provide mechanical support for living organisms and their structures, including most of our buildings and other installations. Soils influence the hydrological processes, including infiltration, percolation, drainage, streamflow, and surface as well as underground water storage. Soils regulate exchange of material, energy, water, and gas within the lithosphere–hydrosphere–biosphere–atmosphere system. Soil is a source and sink of pollutants. Moreover, soil respiration and carbon sequestration may influence climate change. Soil is, without question, critical to the world, supplying virtually all the food and fiber that sustain the human population and providing ecosystem services that support life (Anderson 2010). It is a nonrenewable natural resource in human life time frame (Lal 2009).

Soil is not land itself; it is a part of the land. The total land area of the world is estimated to be 130,575,894 km^2, including rocky surfaces, deserts, ice-covered areas, and lands with soil. There are about 38.5 million km^2 or 29.45 % of the earth's ice-free land surface which is too dry for sustainable human habitation (Beinroth et al. 1994). About 20.2 million km^2 or (15.46 %) of the land occurs in the cold tundra zone, which are not easily amenable to normal agriculture. There are other constraints, which prevent the use of soils for agriculture. For example, saline and alkaline soils occur in 3,105,000 km^2 or 2.4 % of the land surface, and soil acidity affects 18,420,100 km^2 or 14.1 % of the total land (Eswaran et al. 1997). According to Buringh (1989), between 11 and 12 % of the land surface is generally suitable for food and fiber production, 24 % is used for grazing, forests occupy about 31 %, and the remaining 33 % has too many constraints for most uses.

All agricultural soils are not fertile and productive. Some soils are naturally unproductive; some are arid and saline; some are very sandy and dry; and some are wet and waterlogged for a part or most of the growing season. Advanced water management techniques including irrigation and drainage have enabled some use of the dry lands, wetlands, and peatlands. There are sloping lands, sandy soils, and soils with low nutrient-holding capacity. Many soils in desert regions are irrigated, but these are considered unsustainable. Lal (1989) estimates that about 0.5 ha of cropland per capita is needed to sustain the human population at an acceptable level. But there are many countries where the per capita land is less than 0.07 % (Smil 1987). Mismanagement and misuse have degraded many productive lands worldwide. Oldeman et al. (1991) suggest that about 17 % of the global land area is degraded by human interventions. The consequences of land degradation not only affect the performance of the land for food and fiber production but also have grave consequences for the environment. Formation of an inch top soil may need more than thousands of years; so it should not be allowed to degrade through our careless mismanagement.

1.2 Need for Management and Protection of Soil

The European council declared The European Charter on Soil in 1972, emphasizing the need for management and protection of soil resources (Tolgyessy 1993). The Charter states:

(1) Soil is one of the most valuable of man's properties. It enables the life of plants, animals and humans on the earth; (2) Soil is a limited resource which is easily destroyed; (3) The industrialized society employs soil for agriculture as well as for industrial and other purposes. The policy of regional planning must combine considerations of the soil properties, and the present and future needs of the society; (4) Agriculturalists and foresters must apply methods which protect the soil quality; (5) Soil must be protected against erosion; (6) Soil must be protected against pollution; (7) Development of urbanization must be planned so that the neighboring areas are exposed to the minimum possible damage; (8) During the planning of engineering projects the effects on the soil must be evaluated so that the price includes provision for adequate protective measures; (9) A list of soil resources is a prerequisite for any planning; (10) To ensure the extensive use and protection of soil, further research and interdisciplinary cooperation are required; (11) Attention at all professional levels as well as ever-increasing attention of the general public should be paid to soil protection; and (12) Governments and State authorities must carefully plan and nurture and interdisciplinary cooperation are required.

Some lands such as deserts and ice-covered areas do not contain soil. Where there is soil, there is some sort of vegetation – forests, grasslands, or croplands. The natural vegetation in many areas has been removed to use the soil for cropping or some other purposes, including urban and industrial use. Some vegetation has been retained in its original form or disturbed, and some have been exploited, legally or illegally, judiciously or indiscriminately, for our need. The European Charter rightly states that soil is a limited resource. There is only 11–12 % land area suitable for agricultural crop production without much limitation. Not all agricultural soils are fertile and productive. Soils of only about 3 % of the total land area of the earth have a high level of productivity (Aswathanarayana 1999). Therefore, soils need to be managed and protected sustainably so that they can remain productive in future. Figure 1.1 shows different categories of land in the world.

1.3 Land Capability and Land Capability Classification

Land capability is the ability of land to accept a type and intensity of land use permanently, or for a specified period under a certain management without long-term degradation (Houghton and Charman 1986). The land capability denotes the potential of land for use in agriculture, horticulture, forestry, and other uses based on the

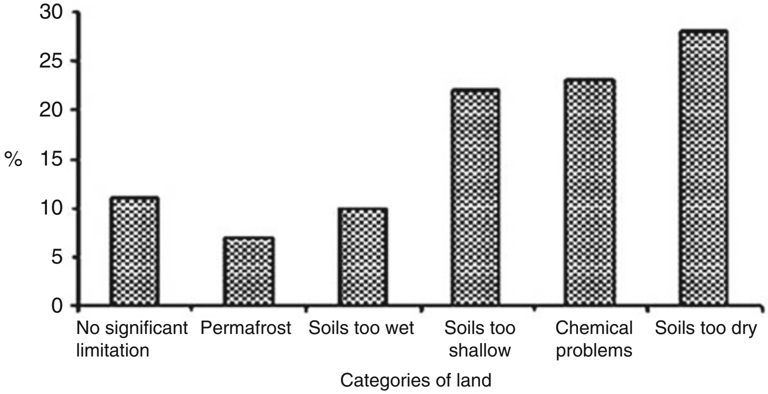

Fig. 1.1 Global soil conditions (Based on data from FAO 1998)

degree of limitation imposed by its biophysical properties. It is based primarily on climate, a number of soil properties, (e.g., depth and stoniness), wetness, erosion risk, and slope.

The USDA Soil Conservation Service devised the land capability classification system (Helms 1977) during the late 1930s and early 1940s. Some form or adaptation of land capability classification is used throughout the world (Olson 1974; FAO 1999). Scientists are continually refining and improving land classification systems (Eswaran et al. 2000; Fischer et al. 2000). Land capability classification is a system of grouping soils primarily on the basis of their capability to produce common cultivated crops and pasture plants without deteriorating over a long period of time. The land capability classification is one of the interpretive groupings made also primarily for agricultural purpose. USDA Nature Conservation Service grouped soils into following 8 land capability classes.

Class I. Class I soils have few limitations and are suited to a wide range of plants; they can be safely used for cultivated crops, pasture, range, woodland, and wildlife. Soils are nearly level with low erosion hazard. Soils are deep, fertile, and responsive to fertilizers.

Class II. Soils of this class have some limitations reducing choice of crops and may require moderate conservation practices. These soils require careful management and conservation to prevent deterioration or to improve air and water relations when the soils are cultivated. The limitations are few and the practices are easy to apply.

Class III. Soils have severe limitations that reduce the choice of plants or require special conservation practices or both. Soils have more restrictions than those in Class II, and when used for cultivated crops, the conservation practices are usually more difficult to apply and to maintain. They may be used for cultivated crops, pasture, woodland, range, or wildlife food and cover.

Class IV. Soils in this class have severe limitations that restrict the choice of plants require very careful management, or both. The restrictions in use for soils in this class are greater than those in Class III, and the choice of plants is more limited. These soils may be used for cultivated crops, pasture, woodland, range, or wildlife food and cover.

Class V. Soils in this class have little or no erosion hazard but have other limitations impractical to remove that limit their use largely to pasture, range, woodland, or wildlife food and cover. Soil limitations restrict the kinds of plants that can be grown. They are nearly level, but some are wet, frequently overflowed, are stony, have climatic limitations, or have some combination of these limitations.

Class VI. Soils have severe limitations that make them generally unsuited to cultivation and limit their use largely to pasture or range woodland, or wildlife food or cover. Physical conditions of soils in this class are such that it is practical to apply range or pasture improvements. Limitations include steep slope, severe erosion hazard, stoniness, and shallow rooting.

Class VII. Soils have very severe limitations that make them unsuited to cultivation and that restrict their use largely to grazing woodland, or wildlife. Limitations include very steep slopes, erosion, shallow soil, stoniness, wet soil, salts, and unfavorable climate.

Class VIII. Soils and landforms in this class have limitations that preclude their use for commercial plant production and restrict their use to recreation wildlife, or water supply or to aesthetic purposes. Limitations include erosion, stoniness, wet soil, low moisture, salts, and severe climate.

1.4 Soil Fertility, Soil Productivity, Soil Quality, and Soil Health

To avoid confusions, let us first define and explain some terms closely related to each other and frequently used in soil science literature. These terms are not only important in soil use but also for its future management and environmental impact.

1.4.1 Soil Fertility

Soil fertility refers to the capacity of soils to supply plant nutrients in available forms, in proper balance and the absence of any sort of toxicity. Plants absorb 14 nutrients from the soil (nitrogen, phosphorus, potassium, calcium, magnesium, sulfur, iron, manganese, copper, molybdenum, zinc, boron, chlorine, and nickel). Plants need the presence of these nutrients in ionic forms (soluble and exchangeable) for absorption. If the soil contains inadequate amount of any one or more of these nutrients, plants will suffer in growth and reproduction. If the soil is adequately fertile, plants may grow satisfactorily or not, depending on the provision of other requirements from the soil, including air, water, and temperature. If the soil is fertile but the soil is not appropriately managed, production of crops may be low. So, fertile soils may or may not be productive depending on other properties and soil management. By management infertile soils may be made fertile and productive.

1.4.2 Soil Productivity

Soil productivity is the ability of soil to give optimum yields under standard set of management practices. Yield is the amount of production of biomass. Yield may be biological (total biomass) or economic (grains, vegetables, wood, fiber, or other products such as resins, sugars, and oils). A poorly fertile soil may be made productive by fertilizer addition, and a dry soil may be made productive by irrigation. A waterlogged soil may be naturally productive for rice, but not for potato. Draining the soil may make it productive for potato, too.

1.4.3 Soil Health

Soil health is defined as the continued capacity of soil to function as a vital living system, by recognizing that it contains biological elements that are key to ecosystem function within land-use boundaries (Doran and Zeiss 2000; Karlen et al. 2001). These functions are able to sustain biological productivity of soil, maintain the quality of surrounding air and water environments, as well as promote plant, animal, and human health (Doran et al. 1996). Soil health is the expression of ability of a soil to meet its range of ecosystem functions as appropriate to its environment. This term is used to assess the ability of a soil to sustain plant and animal productivity and diversity, maintain or enhance water and air quality, and support human health and habitation. It underlies that soil is not just a growing medium; rather it is a living, dynamic, and ever-so-subtly changing environment.

1.4.4 Soil Quality

The soil quality concept evolved in the early 1990s (Doran and Safley 1997; Wienhold et al. 2004), and the first official application of the term was approved by the Soil Science Society of America Ad Hoc Committee on Soil Quality. It was discussed elaborately by Karlen et al. (1997). Soil quality has been defined as “the capacity of a reference soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation." Subsequently the two terms soil quality and soil health are used interchangeably (Karlen et al. 2001), although it is important to distinguish that soil quality is related to soil function (Karlen et al. 2003; Letey et al. 2003), whereas soil health presents the soil as a finite nonrenewable and dynamic living resource (Doran and Zeiss 2000). The quality of soil is rather dynamic and can affect the sustainability and productivity of land use. It is the end product of soil degrading or conserving processes and is controlled by chemical, physical, and biological components of a soil and their interactions (Papendick and Parr 1992).

1.4.4.1 Soil Quality Classes

The concluding remarks of the meeting on "Land Resources: On the Edge of the Malthusian Precipice?" (Greenland et al. 1998), "if all the resources are harnessed and adequate measures taken to minimize soil degradation, sufficient food to feed the population in 2020 can be produced, and probably sufficient for a few billion more," inspired Eswaran et al. (1999) to review the soil quality status of the world. They defined soil resilience and soil performance and divided each into low, medium, and high categories. Combining three soil performance classes and three soil resilience classes, Eswaran et al. (1999) divided world soils into nine land quality (in this classification land and soil are used synonymously) classes. They defined land quality, soil resilience, and soil performance as:

Land Quality: The ability of the land to perform its function of sustainable agriculture production and enable it to respond to sustainable land management.

Soil Resilience: The ability of the land to revert to a near original production level after it is degraded, as by mismanagement. Land with low resilience is permanently damaged by degradation.

Soil Performance: The ability of the land to produce (as measured by yield of grain, or biomass) under moderate levels of inputs in the form of conservation technology, fertilizers, pest, and disease control. The definitions of the land quality classes according to Eswaran et al. (1999) are shown below:

Land quality class	Performance and resilience
Class I	High performance, high resilience
Class II	High performance, medium resilience
Class III	Medium performance, high resilience
Class IV	High performance, low resilience
Class V	Medium performance, medium resilience
Class VI	Low performance, high resilience
Class VII	Low performance, low resilience
Class VIII	Low performance, medium resilience
Class IX	Low performance, low resilience

Table 1.1 Land quality classes and their properties

Land quality	Properties
Class	
I	Class I land is the prime land. The soils are highly productive with few limitations. Moisture and temperature conditions are ideal for annual crops. Soil management consists largely of sensible conservation practices to minimize erosion, appropriate fertilization, and use of best available plant materials. Risk for sustainable grain crop production is generally <20 %
II and III	The soils are good and have few problems for sustainable production. Care must be taken to reduce degradation, particularly for Class II soils. The lower resilience characteristics of Class II soils make them more risky, particularly for low-input grain crop production. However, their productivity is generally very high, and consequently, response to management is high. Conservation tillage is essential, buffer strips are generally required, and fertilizer use must be carefully managed. Due to the relatively good terrain conditions, the land is suitable for national parks and biodiversity zones. Risk for sustainable grain crop production is generally 20–40 %, but risks can be reduced with good conservation practices
IV, V, VI	If there is a choice, these soils must not be used for grain crop production, particularly soils belonging to Class IV. All three classes require important inputs of conservation management. In fact, no grain crop production must be contemplated in the absence of a good conservation plan. Lack of plant nutrients is a major constraint, and so a good fertilizer use plan must be adopted. Soil degradation must be continuously monitored. Productivity is not high, and so low-input farmers must receive considerable support to manage these soils or be discouraged from using them. Land can be set aside for national parks or as biodiversity zones. In the semiarid areas, they can be managed for range. Risk for sustainable grain crop production is 40–60 %
VII	These soils may only be used for grain crop production if there is a real pressure on land. They are definitely not suitable for low-input grain crop production; their low resilience makes them easily prone to degradation. They should be retained under natural forests or range, and some localized areas can be used for recreational purposes. As in Class V and VI, biodiversity management is crucial in these areas. Risk for sustainable grain crop production is 60–80 %
VIII, IX	These are soils belonging to very fragile ecosystems or are very uneconomical to use for grain crop production. They should be retained under their natural state. Some areas may be used for recreational purposes but under very controlled conditions. In Class IX, which is largely confined to the boreal area, timber harvesting must be done very carefully with considerable attention to ecosystem damage. Class VIII is mainly the deserts. Risk for sustainable grain crop production is >80 %

Adapted from Eswaran et al. (1999)

The properties of the different land quality classes are shown in Table 1.1.

1.4.4.2 Assumptions

The classification was intended to be universal and several assumptions were made: (1) It focuses on the inherent ability of soil to produce grain crops in a sustainable manner. Fragile ecosystems, such as wetlands, the cold tundra zone, and deserts are excluded from the assessment. (2) Crop performance and response to management are emphasized. They depend on soil and climatic conditions. (3) Irrigation is not considered in this analysis. (4) Crops other than food crops are not considered. (5) Productivity is a function of management. For purposes of evaluating the response to management of the inherent land quality classes, the levels of input as defined in FAO (1976) are used. (6) Assessment of resilience is empirical, as there are few measurements of resilience available (Greenland and Szabolcs 1994). (7) The integrity of the environment and a rational balance between agricultural use and environmental management (Tinker 1998) are important considerations. (8) A general notion of risk is included. Though some values are given, these are to indicate magnitudes rather than absolute numbers.

Class I lands or prime lands occur in the USA, Argentina, Uruguay, southern Brazil, Europe, northern China, and South Africa. Such lands are not found in the tropics. Tropical soils are generally of low productivity. Class II and III lands are extensive in the tropics and in the temperate areas. Most of these lands are under some form of agriculture and irrigated, when possible, in the semiarid parts. Only about 3.1 % of the global land surface can be considered as prime or Class I land. Along with Class II and III lands, the 12.6 % of land represent the total land area that is generally free of constraints for most agricultural uses. They are not equally spread around the globe. A larger portion of these soils are in the temperate countries of the world. Class I and II lands generally have good resilience and are highly productive. They are benefitted from conservation technologies that are directed to preserving the favorable attributes. Many countries have very few, none of either Class I or II lands, or have utilized all available I, II, and, III lands and consequently have to use more inferior lands. A major part of Class IV, V, and VI lands, particularly the Class V lands, are in the tropics. In the Amazon basin, Central Africa, and Southeast Asia, these form large tracts of forests. Class IV, V, and VI lands occupy a significant part of the earth's surface (36.8 million km^2) and also support more than 50 % of the world's population. Class IV or poorer quality lands may be managed with high inputs and can be made productive (Buol and Eswaran 1994). From a sustainability point of view, Classes VII, VIII, and IX are not suitable for agriculture.

Eswaran et al. (1999) estimated that the total global land area suitable for cultivation is 60.2 million km^2. WRI (1997) estimated that about 49.77 million km^2 were under cultivation in 1995. This suggests that the amount of globally available land is only about 10 million km^2 for future cropping, forestry, and for other uses. The percentage of cropland in Asia, Africa, South America, North America, and Europe are 15.2, 6.3, 6.0, 13.0, and 6.0, respectively. The proportions of pasture in the corresponding regions are 20, 22, 14, 08, and 23 % (WRI 1997). Figure 1.2 shows area of land under different quality classes.

1.5 Global Arable Land

The world population up to July 2009 became 6.79 billion, and the global arable land area is estimated as 1.351 billion hectares (Anonymous 2009). Arable land per capita on a global basis stands at only 0.20 ha, and according to Eswaran et al. (1999), per

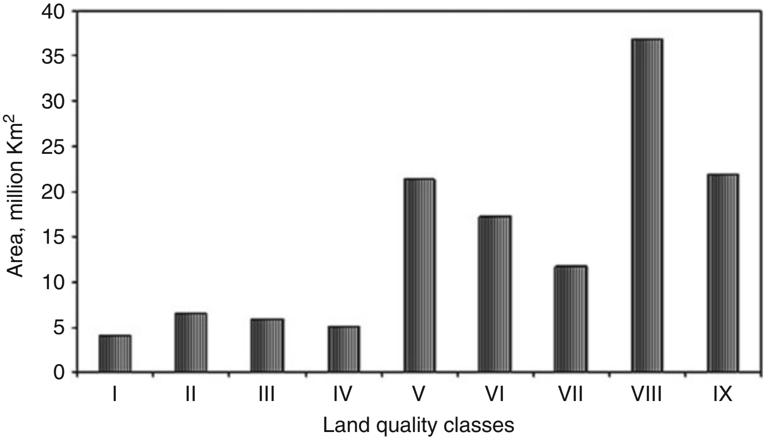

Fig. 1.2 Area of land under different quality classes (Data from Eswaran et al. 1999)

Table 1.2 Distribution of arable lands in different continents

Continents	Percent of global arable land	Percent of global population	Per capita arable land (hectare)
Asia	31.94	56.7	0.11
North America	17.09	6.7	0.52
Africa	14.16	14.2	0.20
Europe	11.31	8.8	0.26
Eurasia	10.72	3.2	0.68
South America	7.88	5.8	0.27
Australia	3.47	0.30	2.23

The global arable land area is estimated as 1.351 billion hectares, and the global population is estimated as 6.790 billion people (Anonymous 2009)

capita productive land less than 0.5 ha is not sufficient. Asia has the largest share of global arable land (32 %) followed by North America at 17 % and Africa at 14 %. Africa and Asia together have 46 % of total arable land, 71 % of the global population and the lowest amount of arable land per capita. Every year, 0.3–0.8 % of global arable land is rendered unsuitable for agricultural production due to soil degradation and wind and water erosion accounts for 84 % of this degradation (den Biggelaar et al. 2004a, b). Almost 45 % of arable soils worldwide are now affected by some form of degradation (Lal 2007). The global distribution of arable land is given in Table 1.2.

1.6 Classification of World Soils

Two major classification systems deal with soils of the world – the World Reference Base for Soil Resources and the Soil Taxonomy. The World Reference Base for Soil Resources (WRB) was developed by the Food and Agriculture Organization of the

United Nations (FAO 1998) in conjunction with the International Union of Soil Sciences and the International Soil Reference and Information Center (ISRIC) at Wageningen University, the Netherlands. Soil Taxonomy was developed through several revisions over the past five decades by the US Department of Agriculture (USDA 1975). In this hierarchical system of Soil Taxonomy, soil order is the highest category which is divided into suborders, suborders into great groups, and so on. There are 12 orders, 64 suborders, 319 great groups, 2,484 subgroups, 8,000 families, and an unknown number of soil series. Characteristics of the soil orders and their suborders are briefly described below.

1.6.1 Alfisols

Alfisols are fine-textured soils with high content of exchangeable bases. They have mostly developed in the humid temperate and also in the humid tropical regions under deciduous forests. They have accumulation of clay in the B horizon to form argillic, kandic, or natric horizon, with BSP >35 % in the lower part or below the argillic or kandic horizon. They may contain petrocalcic horizons and duripan, fragipan, and plinthite. There is enough moisture for plant growth for three consecutive months during the growing seasons. There is relatively little accumulation of organic matter in mineral horizons. These soils are highly fertile and are extensively cultivated with widely diverse cropping patterns as favored by climatic conditions. Some are used for hay, pasture, range, and forests. Alfisols comprise 9.6 % of the ice-free land of the earth. (The WRB equivalents of Alfisols are Albeluvisols, Planosols, Luvisols, and Lixisols.) Alfisols have five suborders. They are:

Aqualfs: Aqualfs are Alfisols that have aquic conditions (shallow groundwater table that saturates soil with water) for some time in most years within 50 cm of the mineral horizon and redoximorphic features in the upper 12.5 cm of the argillic, natric, or kandic horizon. Aqualfs are abundant in humid regions and are primarily used for rice cultivation. They are fairly fertile, and other crops including corns (maize) and soybeans can be grown if artificially drained. Nearly all Aqualfs are believed to have supported forest vegetation in the past.

Cryalfs: Cryalfs are more or less freely drained Alfisols of the cold regions (cryic soil temperature regime) and occur mostly at high elevations, as in the Rocky Mountains in the Western USA. They normally have a udic moisture regime. Most of the Cryalfs are used as forest because of their short, cool growing season.

Udalfs: Udalfs are the more or less frequently drained Alfisols that have udic soil moisture regime and a frigid, mesic, isomesic, or warmer temperature regime. Udalfs are very extensive in the USA and in Western Europe. Most Udalfs with a mesic or warmer temperature regime have or had deciduous forest vegetation, and many of the frigid temperature regimes have or had mixed coniferous and deciduous trees.

Ustalfs: Ustalfs have an ustic soil moisture regime and a frigid, mesic, isomesic, or warmer temperature regime. Ustalfs are the Alfisols of subhumid to semiarid regions. They occur in the USA, Africa, India, South America, Australia, and Southeastern Asia. Sorghum, wheat, and cotton are commonly cultivated with irrigation.

Xeralfs: Xeralfs have xeric soil moisture regime common of regions that have Mediterranean climate. They are found in South Africa, Chile, Western Australia, Southern Australia, and the Western USA. They are dry for extended periods in summer, but enough is available in winter. Small grains and other annuals are common crops without irrigation. Grapes and olives are also common crops where the climate is thermic. With irrigation, a wide variety of crops can be grown.

1.6.2 Andisols

Andisols are characterized by andic materials. Andic materials include volcanic ash, pumice, and cinders deposited during volcanic eruptions. These materials undergo transformation to amorphous or poorly crystallized silicate minerals, including allophane, imogolite, and ferrihydrite. Andisols are young soils and have not had enough time to be highly weathered. These soils are fine-textured and have a high content of fresh weatherable minerals and a high cation exchange capacity. They may also contain considerable organic matter as aluminum–humus complex. They have low bulk densities. These soils are widely distributed in all geographical regions near sources of volcanoes. Andisols are generally fertile and are used for agriculture unless restricted by slope, altitude, soil moisture and temperature regimes, etc. One of the most important characteristics of Andisols is their high capacity to fix phosphorus on the surface of the amorphous minerals (Cordova et al. 1996). This is perhaps the principal chemical constraint of Andisols. Some Andisols are left under tundra and forests. Andisols cover more than 124 million hectares, or approximately 0.7 % of the earth's surface. Major areas of Andisols include Chile, Peru, Ecuador, Columbia, Central America, the USA, Kamchatka, Japan, the Philippines, Indonesia, and New Zealand. (The WRB equivalent of Andisols are the Andosols.) Andisols have seven suborders. They are:

Aquands: Aquands are the Andisols with aquic conditions at or near the surface. These soils have dark-colored surface horizons that meet the requirements for a histic, umbric, or mollic epipedon. Aquic conditions result in redoximorphic features. Aquands occur locally in depressions and along floodplains where water tables are at or near the soil surface for at least part of the year.

Cryands: Cryands are defined as Andisols with cryic soil temperature regimes. These soils are the Andisols of high latitude (e.g., Alaska, Kamchatka) and high altitude (e.g., Sierra Nevada in the USA). They are usually occupied by cold tolerant forests.

Torrands: Torrands are more or less well-drained Andisols of dry regions. They have an aridic/torric soil moisture regime and a frigid or warmer soil temperature regime. Natural vegetation is mostly desert shrubs. These soils are not extensive occurring mostly in the western part of North America, Hawaii, or other Pacific regions. Most of the soils formed under grass or shrub vegetation.

Udands: Udands are more or less well-drained Andisols of moist regions. They have a udic soil moisture regime. Udands are Andisols of the humid climates. They are the most extensive Andisols. These soils are moderately extensive on the Pacific Rim, including Washington, Oregon, and Hawaii in the USA. Most Udands formed under forest vegetation.

Ustands: Ustands are defined as Andisols with ustic soil moisture regimes. These soils are distributed in intertropical regions that experience seasonal precipitation distribution. They are found mostly in Mexico, Western USA, Pacific Islands, and the eastern part of Africa. Most Ustands are formed under grass, shrub, or forest vegetation.

Vitrands: Vitrands are relatively young Andisols that are coarse-textured soils and are dominated by volcanic glass. Most Vitrands are found near volcanoes. Vitrands are abundant in Oregon, Washington, and Idaho of the USA where they form mainly under coniferous forest vegetations. They are the Andisols that have a frigid or warmer soil temperature regime. They have a low water-holding capacity. Vitrands are restricted to ustic and udic soil moisture regimes.

Xerands: Xerands are more or less well-drained Andisols that have a xeric soil moisture regime and a frigid, mesic, or thermic temperature regime. They are temperate Andisols with very dry summers and moist winters. Most Xerands formed under coniferous forest vegetation and some formed under grass or shrub vegetation.

1.6.3 Aridisols

Aridisols are soils of the arid regions including cold polar, cool temperate, and warm deserts. Aridisols may also occur in semiarid areas outside the zones broadly classified as arid, for example, in local conditions imposing aridity such as steep, south-facing slopes in the northern hemisphere and in soils whose physical properties limit water infiltration or favor excessive drainage. Aridisols are classified on the basis of their soil moisture regime which is dry in all parts >50 % of the time in most years and not moist for as much as 90 consecutive days when the soil is warm enough (>8 °C) for plant growth. In an aridic/torric soil moisture regime, potential evapotranspiration greatly exceeds precipitation during most of the year. In most years, little or no water percolates through the soil. This hydrologic regime has a distinctive influence on the development of such soils. However, shifting sands of deserts are not included in Aridisols.

There is low chemical weathering, low leaching, and sparse plant growth. Soil organic matter content is low so that ochric horizons abundantly develop in Aridisols. Salts (chlorides, sulfates, carbonates) released by limited chemical weathering are not usually translocated to considerable depths but are accumulated on the surface and, where there is some downward movement of water, in the B horizon. Aridisols have one or more of the following within 100 cm of surface a calcic, cambic, gypsic, natric, petrocalcic, petrogypsic, or salic horizon. An argillic horizon is found in some Aridisols. This is believed to have developed under a moister climate of the past. Aridisols are sparsely vegetated, mostly in xeric shrub lands with xerophytes, cactus, and thorns. They may be cultivated if irrigation can be given, but source of irrigation water is also scanty there. Aridisols comprise about 12 % of the world's ice-free land surface. (The WRB equivalents of Aridisols are Durisols, Gypsisols, and Solonchaks.) Aridisols have seven suborders. They are:

Cryids: Cryids are the Aridisols of the cold climates. These soils are characteristically developed at high elevations, dominantly in the mountain and basin areas of the USA and Asia and other parts of the world. Cryids commonly show evidence of periglacial features.

Salids: Salids are Aridisols with accumulations that are more soluble than gypsum. The most common form is sodium chloride, but sulfates and others may also occur. These soils are common in depressions in the deserts or in closed basins in wetter areas bordering deserts. Some salts may be brought to the upper horizons by capillary rise of groundwater.

Durids: Durids are the Aridisols that have an accumulation of silica. There is a duripan which is cemented partly with opal or chalcedony. The soils commonly have calcium carbonate. The duripan restricts movement of water and penetration of roots. These soils occur in the western part of the USA particularly in Nevada. They are not known to occur outside the USA.

Gypsids: Gypsids are the Aridisols that have an accumulation of gypsum. These soils occur in Iraq, Syria, Saudi Arabia, Iran, Somalia, West Asia, and in some of the most arid regions of the USA. When the gypsic horizon occurs as a cemented impermeable layer, it is recognized as the petrogypsic horizon.

Argids: Argids are the Aridisols that have accumulation of clay. These soils have an argillic or natric horizon. The presence of an argillic horizon is commonly attributed to a moister paleoclimate. Most Argids occur in North America with a few recognized in the deserts of North Africa or the Near East.

Calcids: Calcids are the Aridisols that have accumulation of residual calcium carbonate or was added as dryfall. Precipitation is inadequate to leach or move the carbonates to great depths. These soils are extensive in the western USA and other arid regions of the world.

Cambids: These are the Aridisols with the least degree of soil development. They have a cambic horizon that has its upper boundary within 100 cm of the soil surface. These soils are the most common Aridisols in the USA and other parts of the world.

1.6.4 Entisols

Entisols are defined as soils that have little or no sign of horizon differentiation. Most Entisols are basically unaltered from their parent materials. Actually they are affected to a limited extent by translocation processes. However, there is considerable darkening of the surface soil by organic matter. The presence of unweatherable parent materials, removal of soil materials by continuous erosion, continuous deposition of silts with floodwater in active floodplains, cold and dry climates, and insufficient time after rock exposure or sediment deposition are the causes of delayed soil development in Entisols. These soils are distributed over a wide geographic area and can be found in any climate and under any vegetation. Entisols along river floodplains are often intensively farmed and are some of the most agriculturally productive soils in the world. Most Entisols are used for pasture, rangeland, and forests. Entisols occupy about 16 % of the global ice-free land surface. (The WRB equivalents of Entisols are Anthrosols, Arenosols, Fluvisols, Regosols, Stagnosols, and Umbrisols.) Entisols have four suborders. They are:

Aquents: These are the wet Entisols. They may be found in tidal marshes, on deltas, on the margins of lakes where the soils are continuously saturated with water, on floodplains along streams where the soils are saturated at some time of the year, or in areas of wet, sandy deposits. Many Aquents have gleying with bluish or grayish colors and redoximorphic features. They may have any temperature regime. Most are formed in recent sediments and support vegetation that tolerates permanent or periodic wetness. Vast areas of alluvial Aquents are used for rice cultivation in South and Southeast Asia, including Bangladesh. Some Aquents have sulfidic materials (former acid sulfate soils).

Arents: Arents are the Entisols that do not have horizons because they have been deeply mixed by plowing, spading, or other methods of moving by humans. Arents may have 3 % or more, by volume, fragments of diagnostic horizons in one or more subhorizons at a depth between 25 and 100 cm below the soil surface.

Fluvents: Fluvents are mostly brownish to reddish soils that are formed in recent alluvial sediments, mainly on floodplains, fans, and deltas of rivers and small streams but not in back swamps where drainage is poor. Strata of clayey or loamy materials commonly have more organic carbon than the overlying, more sandy strata. Fluvents are often found associated with Aquents in floodplains. Rice and jute are grown in many Fluvents.

Psamments: Psamments are Entisols that are very sandy at all layers. Some Psamments form in poorly graded but well-sorted sands on shifting or stabilized deposits, in cover sands, or in sandy parent materials that were sorted in an earlier geologic cycle. Psamments occur under any climate without permafrost within 100 cm of the soil surface. They can have any vegetation and can be cropped with irrigation. Psamments on old stable surfaces commonly consist of quartz sand. These soils are poorly fertile and dry and often show nutrient deficiencies.

1.6.5 Gelisols

Gelisols are soils that contain gelic materials (mineral or organic soil materials that show cryoturbation, cryodesiccation, and/or ice segregation in the active layer). Gelisols are soils of very cold climates that contain permafrost within 2 m of the surface. Freezing and thawing in the active layer influence soil formation in Gelisols. Permafrost restricts the downward movement of water. Thus, there are few diagnostic horizons in Gelisols, if any. Cryoturbation results in irregular or broken horizons, organic matter accumulation on the permafrost table, oriented rock fragments, and silt caps on rock fragments. These soils are limited geographically to the high-latitude polar regions and localized areas at high mountain elevations. Gelisols are the permafrost-affected soils that occur throughout the zone of continuous permafrost in Antarctica (Bockheim 1995). (Gelisols are called Cryosols in WRB classification system.) Gelisols occur in 8 % of the earth surface. Gelisols have three suborders. They are:

Histels: Histels have organic horizons similar to Histosols except that they have permafrost within 2 m below the ground. They have 80 % or more organic materials from the soil surface to a depth of 50 cm or to a glacic layer or densic, lithic, or paralithic contact, whichever is shallower. These soils occur predominantly in subarctic and low Arctic regions of continuous or widespread permafrost. The natural vegetation in Histels is mostly mosses, sedges, and shrubs. The soils are used as wildlife habitat.

Turbels: Turbels are Gelisols that commonly show cryoturbation and contain tongues of mineral and organic horizons, organic and mineral intrusions, and oriented rock fragments. Organic matter is accumulated on top of the permafrost, and ice wedges are common features in Turbels.

Orthels: Orthels are soils that show little or no cryoturbation (less than one-third of the pedon). These soils occur primarily within the zone of discontinuous permafrost, in alpine areas where precipitation is greater than 1,400 mm per year. The natural vegetation is mostly lichens, mosses, sedges, shrubs, black spruce, and white spruce. The soils are used mostly as wildlife habitat. They occur throughout the Gelisol area in Alaska. The vegetation is mostly mosses, sedges, shrubs, and black spruce.

1.6.6 Histosols

Histosols are permafrost-free soils dominated by organic soil materials. Organic soil materials consist of organic debris accumulating at the surface in which the mineral component does not significantly influence the properties of soils. Organic soil materials have either:

1. Under water saturated conditions 18 % organic carbon (30 % organic matter) or more if the mineral fraction has 60 % or more clay, or 12 % organic carbon (20 % organic matter) if the mineral fraction has no clay, or a proportional intermediate organic carbon for intermediate content of clay

2. If never saturated with water for more than a few days, 20 % or more organic carbon

Histosols typically form in settings where poor drainage inhibits the decomposition of plant and animal remains, allowing these organic materials to accumulate over time. Thus, they have developed in organic parent materials, and they are mostly soils that are commonly called bogs, moors, or peats and mucks. Peat is the name given to slightly decomposed organic material in soil, while muck is used for the rotten, highly decomposed material. The peat is used for fuel, potting soil in greenhouses, and for packing. Histosols can be cultivated only if artificially drained. Histosols serve as important habitats for wetland plants and animals and as carbon reservoirs. Histosols are ecologically important because of the large quantities of carbon they contain. Typically, Histosols have very low bulk density (Chap. 5) and are poorly drained because of their occurrence in low-lying areas and high organic matter content. Most Histosols are acidic, and many are deficient in plant nutrients. Many Histosols are not suitable for cultivation because of poor drainage and low chemical fertility. However, many other Histosols formed on recent glacial lands can be very productive when drained. They can sometimes be used for orchards and vines if carefully managed. However, there is a great risk of wind erosion, shrinkage, subsistence, and compaction. (The same Histosols name is given in WRB.) Histosols occupy only 1.2 % of the global ice-free land surface.

Histosols have four suborders mostly distinguished on the basis of the state of organic matter and drainage. They are:

Fibrists: Fibrists are the wet, slightly decomposed Histosols. The largest extent is in southern Alaska of the USA. Most of these soils support natural vegetation of widely spaced, small trees, shrubs, and grasses.

Folists: Folists are the more or less freely drained Histosols that consist primarily of horizons derived from leaf litter, twigs, and branches resting on bedrock or on fragmental materials. Most of these soils support forest vegetation. Some of the soils mainly support grass. A few of the soils are used for specialty crops or for urban or recreational development.

Hemists: Hemists are the wet Histosols in which the organic materials are moderately decomposed. They are extensive in Minnesota and Alaska. Most Hemists support natural vegetation and are used as woodland, rangeland, or wildlife habitat. Some have been cleared and drained and are used as cropland.

Saprists: Saprists are the wet Histosols in which the organic materials are well decomposed. The largest extent in the USA is in Michigan, Florida, Wisconsin, Minnesota, and Alaska. Small areas are common on the Atlantic and gulf coasts. Many Saprists support natural vegetation and are used as woodland, rangeland, or wildlife habitat. Some of the soils, mostly those with a mesic or warmer temperature regime, have been cleared and drained and are used as cropland.

1.6.7 Inceptisols

Inceptisols are soils that exhibit only the beginning of soil profile development. They are weakly developed soils in that they have minimal horizon differentiation. They are more developed than Entisols and lack many characteristics of mature soils. Inceptisols may have many kinds of diagnostic horizons except argillic, natric, kandic, spodic, and oxic horizons. The most common horizon sequence is an ochric epipedon over a cambic horizon, with or without an underlying fragipan. Inceptisols typically have a cambic horizon, but one is not required if the soil has a mollic, umbric, histic, or plaggen epipedon or if there is a fragipan or duripan or any placic, calcic, petrocalcic, gypsic, petrogypsic, salic, or sulfuric horizon. Inceptisols are soils of humid and subhumid regions. Inceptisols are widely distributed and occur under a wide range of environmental settings. They are often found on fairly steep slopes, young geomorphic surfaces, wet sites, and on resistant parent materials. Inceptisols occupy 9.9 % of the global ice-free land surface. (The WRB equivalents of Inceptisols are Cambisols, Gleysols, and Umbrisols.) Inceptisols have six suborders. They are:

Anthrepts: Anthrepts are more or less freely drained Inceptisols that have either an anthropic or plaggen epipedon. Most have a cambric horizon. Anthrepts can have almost any temperature regime and almost any vegetation. Anthrepts are usually cultivated soils, but some Anthrepts have been diverted to other land uses.

Aquepts: Aquepts are the wet Inceptisols. The water table remains at or near the surface for much of the year. Most Aquepts have formed in depressions, on nearly level plains, or on floodplains. Aquepts may have almost any particle-size class except fragmental. Many Aquepts in floodplains are used for cultivation of rice.

Cryepts: Cryepts are Inceptisols of the cold regions such as high mountains or high latitudes. They do not have permafrost within 100 cm of the soil surface. They may be formed in loess, drift or alluvium (Chap. 3), or in solifluction (mass wasting of water saturated soil material down the slope, over impermeable surface) deposits. Cryepts occur in the USA in the high mountains of the West, southern Alaska, as well as in other mountainous areas of the world. Vegetation is mostly conifers or mixed conifers and hardwood forests. Few soils are cultivated.

Udepts: Udepts are mainly the more or less freely drained Inceptisols that have a udic or perudic soil moisture regime. They are Inceptisols of humid climates. They are found on nearly level to steeply sloping surfaces. Most of the soils were originally covered with forest vegetation, with some shrubs or grasses. The Udepts of the USA are most extensive in the Appalachian Mountains, on the Allegheny Plateau, and on the west coast. Many Udepts are now under cropping.

Ustepts: Ustepts are mainly the more or less freely drained Inceptisols that have an ustic soil moisture regime. Rainfall occurs mainly during the summer. Some Ustepts are found in older deposits on steep slopes. Native vegetation is commonly grass but some supported trees. Most are used as cropland or pasture.

Xerepts: Xerepts are mainly more or less freely drained Inceptisols that have a xeric soil moisture regime. They are Inceptisols of the temperate regions with very dry summers and moist winters. Xerepts are moderately extensive in the USA and are the most common in California, Oregon, Washington, Idaho, and Utah. The vegetation commonly is coniferous forest on soils with frigid or mesic temperature regimes and shrubs, grass, and widely spaced trees on the soils with a thermic temperature regime.

1.6.8 Mollisols

Mollisols are dark-colored, base-rich mineral soils of the grasslands. They have a mollic epipedon. They may have an argillic, natric, or calcic horizon or an albic horizon. Some have a duripan or a petrocalcic horizon. Mollisols do not have permafrost, organic soil materials, and a spodic horizon. Mollisols may have any of the defined temperature regimes. Mollisols can have any soil moisture regime, but enough available moisture to support perennial grasses seems to be essential. Mollisols are used mainly for small grain in the drier regions and corn (maize) or soybeans in the warmer, humid region. Mollisols comprise 6.9 % of the ice-free land of the earth. (Mollisols are called Chernozems, Phaeozems, and Kastanozems in WRB classification system.) The suborders of Mollisols are:

Albolls: Albolls are the Mollisols that have an albic horizon and fluctuating groundwater table. Most of these soils are saturated with water to or near the soil surface at some time during winter or spring in normal years. These soils developed mostly on broad, nearly level to sloping ridges, on back slopes, or in closed depressions. Most Albolls have developed under grass or shrub vegetation.

Aquolls: Aquolls are the Mollisols that are wet and that have an aquic soil moisture regime. In these soils, the water table remains at or near the surface for much of the year. They have developed under grasses, sedges, and forbs, but a few have had forest vegetation. In the USA, Aquolls are most extensive in glaciated areas of the mid-western states where the drift was calcareous.

Cryolls: Cryolls are more or less freely drained Mollisols of the cold region. They are abundant in the high mountains of the Western USA, on the plains and mountains of Eastern Europe and in Asia. The vegetation of the Cryolls on the plains was mostly grasses. Cryolls in the mountains have either forest or grass vegetation. Cryolls in Alaska support spruce, birch, and aspen trees.

Rendolls: Rendolls are shallow Mollisols over calcareous parent materials such as limestone, chalk, and drift composted of limestone or shell bars of humid regions. These soils are extensive in some parts of the world and formed under forest vegetation or under grass and shrubs.

Udolls: Udolls are Mollisols of humid climates mainly under tall grass prairie (an extensive, level or slightly undulating, treeless tract of land covered with coarse grasses) vegetation, but some could have supported boreal forests (Chap. 14) several thousand years ago. Most of these soils occur in the eastern part of the Great Plains.

Ustolls: Ustolls are Mollisols of semiarid and subhumid climates having an ustic soil moisture regime. Rainfall occurs mainly during a growing season, often in heavy showers, but is erratic. Drought is frequent and sometimes may be severe. Natural vegetation in Ustolls may be grass in the Great Plains and forest in the mountains of Western USA.

Xerolls: Xerolls are the temperate Mollisols with very dry summers and moist winters within a Mediterranean climate. Xerolls have a xeric soil moisture regime. Xerolls are extensive in parts of Turkey, northern Africa near the Mediterranean and in some of the southern republics of the former USSR, and in several states in the USA.

1.6.9 Oxisols

Oxisols develop under a climate characterized by small seasonal variation in soil temperature and no seasonal soil freezing and high annual precipitation. They may have a wide range of soil moisture regimes from aridic to perudic. Oxisols with aridic soil moisture regimes are often considered as paleosols. Usually, Oxisols develop under climatic conditions where precipitation exceeds evapotranspiration for some periods of the year to facilitate the removal of soluble weathering products and favors the residual concentration of kaolinite and sesquioxides, which are essential to form an oxic horizon. Oxisols have the upper boundary of an oxic horizon and no kandic horizon within 150 cm or 40 % or more clay by weight in the fine-earth fraction. Oxisols do not have either of the following: (a) permafrost within 100 cm of the soil surface, or gelic materials within 100 cm of the soil surface, and (b) permafrost within 200 cm of the soil surface. Oxisols consist mainly of quartz, kaolinite, oxides of Fe, Mn, and Al, and organic matter. Oxisols are poorly fertile weathered soils occurring on gentle slopes of geologically old surfaces in tropical and subtropical regions. The natural vegetation ranges from tropical rainforests to desert savannas. Although many Oxisols are extremely infertile, some Oxisols may be made productive when cultivated with appropriate management. Oxisols comprise 7.5 % of the global ice-free land surface. (The WRB equivalents of Oxisols are Ferralsols, Plinthosols, and Nitisols.) Oxisols have five suborders. They are:

Aquox: Aquox are the Oxisols that have a water table at or near the surface for much of the year in shallow depressions and in seepage areas at the base of slopes. There is a tendency to accumulate iron in the form of secondary nodules, concretions, and plinthite.

Perox: Perox are well-drained Oxisols with a perudic soil moisture regime. They are found in continuously humid climates, where precipitation exceeds evapotranspiration in all months.

Torrox: Torrox are the Oxisols of the arid region. They have an aridic (torric) soil moisture regime. Torrox may become productive soils for a variety of crops if water and fertilizers are applied. They occur mainly in Southern Africa, Hawaii, and some areas of Australia.

Udox: Udox are well-drained Oxisols with a udic soil moisture regime. These soils develop in humid areas. There is usually adequate rainfall in normal years to allow for continuous crop growing. Udox occur mostly in South America and in parts of Africa and Asia.

Ustox: Ustox are the Oxisols that have an ustic soil moisture regime. These soils are found in semiarid and subhumid climates. There is at least 90 consecutive dry days in normal years. Available soil moisture is then very low, and cropping is not done in that time. One crop may be grown in the season when rainfall occurs. Two crops may be grown with irrigation in some areas. Ustox occur over a large portion of the interior of South America and in extensive areas of Africa.

1.6.10 Spodosols

Spodosols typically form in coarse-textured parent materials and have a reddish-brown spodic horizon beneath a light-colored E horizon. Sometimes there is a fragipan or another sequum (a sequum is a couplet of an eluvial horizon above an illuvial horizon, usually an E and an underlying B horizon) (Schaetzl and Anderson 2005); many soil profiles in humid regions have an E–B sequum. Those soils that have two sequa are termed bisequal soils (Schaetzl 1996) that has an argillic horizon below the spodic horizon. Some Spodosols have a placic horizon either on or within a spodic horizon or on a fragipan. Some Spodosols have thicker layers than a placic horizon that are cemented by spodic materials (humus–aluminum–clay complex) and organic matter. The particle-size class is mostly sandy, sandy-skeletal, coarse loamy, loamy skeletal, or coarse silty. Spodosols are most extensive in areas of cool, humid, or perhumid climates. They may also form, however, to a limited extent, in warm, humid tropical regions, where they occur mostly in areas of quartz-rich sands with fluctuating groundwater table. Most Spodosols in cool temperate regions are covered with coniferous or, less commonly, hardwood forests. Plenty of Spodosols are found in boreal forest regions. Some have been cleared for agriculture. Spodosols are naturally infertile, but some Spodosols may be made productive by good management. Spodosols occupy 2.6 % of the global ice-free land surface. (Spodosols are called Podzols in WRB system.) Spodosols have four suborders. They are:

Aquods: Aquods are Spodosols that have an aquic soil moisture regime. They are poorly drained soils with a water table at or near the surface for much of the year. A wide variety of hydrophytic (water loving) plants, ranging from sphagnum in cold areas to palms in the tropics, grow on these soils.

Cryods: Cryods are Spodosols that have a cryic soil temperature regime. They are found in high latitude or high elevations. They are abundant in Alaska, in the mountains of Washington and Oregon of the USA, and Canada. Natural vegetation is mostly coniferous forest or alpine tundra.

Humods: Humods are the relatively freely drained Spodosols that have a large accumulation of organic carbon in the spodic horizon. These soils have developed under coniferous forests, and in western Europe, commonly found in sandy materials where heather (a shrubland characterized by open, low growing woody vegetation) is dominant. In the tropics, most Humods have supported a rain forest.

Orthods: Orthods are the relatively freely drained Spodosols that have a moderate accumulation of organic carbon in the spodic horizon. They are most extensive in the Northeastern USA and the Great Lakes States. Most Orthods are used as forest or have been cleared and are used as cropland or pasture. Orthods are naturally infertile, but they can be highly responsive to good management.

1.6.11 Ultisols

Ultisols are red to yellow soils that are quite acidic, often having a pH of less than 5, and that develop in humid tropical (some in temperate) areas under forest vegetation. They are highly weathered soils and have an argillic or a kandic horizon with low base saturation, less than 35 % BSP (by summation of exchangeable bases). The low base saturation status is mainly due to formation in parent material high in silica but low in bases. In some soils, the low base status results from intense leaching of parent material, while in others, a low base status and small quantities of weatherable minerals were initial parent material characteristics. They may have any soil temperature regime and any soil moisture regime except aridic. Leaching is high and bases released by weathering usually are removed by leaching. The red and yellow colors result from the accumulation of iron oxide which is highly insoluble in water. Kaolinite, gibbsite, and aluminum interlayered clays are common in the clay fraction. Major nutrients, such as calcium and potassium, are typically deficient in Ultisols. They are poorly fertile soils which may not be productive for most crops without addition of lime and fertilizers. Ultisols occupy 8.5 % of the global ice-free land surface. (The WRB equivalents of Ultisols are Acrisols, Alisols, and Plinthosols.) Ultisols have five suborders. They are:

Aquults: Aquults are the Ultisols that have a water table at or near the surface for much of the year. Aquults are found extensively on the coastal plains of the USA, particularly on the Atlantic and Gulf of Mexico. Most of the soils are forested.

Humults: Humults are freely drained Ultisols rich in organic matter (> 0.9 % or more organic carbon in the upper 15 cm of the argillic or kandic horizon) of mid or low latitudes. Rainforests are the usual natural vegetation.

Udults: These are more or less freely drained Ultisols that have a udic soil moisture regime. They develop in humid areas with well-distributed rainfall. Most of these soils have a forest vegetation, but some have a savanna.

Ustults: These are freely drained Ultisols that have an ustic soil moisture regime and a relatively low content of organic carbon. These soils are generally found in semiarid and subhumid climates. The vegetation commonly consists of forest or savanna plants.

Xerults: Xerults are freely drained Ultisols that have a xeric soil moisture regime. They are found in areas with very dry summers and moist winters typically of Mediterranean or temperate climates. Natural vegetation consisted mostly of coniferous forest plants.

1.6.12 Vertisols

Vertisols are clayey soils that have deep, wide cracks for a considerable time of the year and have slickensides (a shiny surface of the cracks produced in soils containing a high proportion of swelling clays) within 100 cm of the mineral soil surface. They shrink when dry and swell when moistened. They are generally sticky in the wet season and hard in the dry season. Most Vertisols have an ustic soil moisture regime; some have an aridic and a udic regime. Vertisols generally have 50–70 % clay with a relatively large proportion of fine clay in the clay fraction. The clays in Vertisols consist predominantly of 2:1 and 2:2 layer clay minerals, but some have considerable amounts of other clay minerals. The natural vegetation is predominantly grass, savanna, open forest, or desert shrub. Most Vertisols are well suited to farming if there is plenty of rainfall or irrigation water and if suitable management practices are followed. Because of the low permeability and tendency to remain waterlogged for long periods, Vertisols are often considered as problem soils (Chap. 11). Vertisols are extensive in some parts of the world. They were known as black cotton soils in India. Vertisols occupy 2.4 % of the global ice-free land surface. (Vertisols have the same name in WRB.) Vertisols have six suborders. They are:

Aquerts: Aquerts are the Vertisols that have aquic soil moisture regime. They have a water table at or near the surface for much of the year but are also dry enough for periods for cracks to open. They are found in low areas such as glacial lake plains, floodplains, stream terraces, and depressions.

Cryerts: Cryerts are the Vertisols that have a cryic soil temperature regime. They are soils of the cold climate. They are fine-textured soils and periodically shrink and swell, forming cracks that commonly open in late summer. Cryerts occur on the cold prairies of Canada where they are commonly derived from lacustrine deposits. They also occur in the US Rocky Mountains.

Torrerts: Torrerts are the Vertisols of arid climates. Their cracks commonly stay open for most of the year but may close for at least a few days during rains. Many of these soils are found in closed depressions that may be ponded from time to time by runoff from higher areas. Some Torrerts are found in the southwest of the USA. These soils are commonly used for rangeland.

Uderts: Uderts are the Vertisols of humid areas. They have a udic soil moisture regime. The cracks may not open completely some years due to high precipitation. In the USA, the soils occur on gentle slopes and are derived dominantly from marine shales, marls, and alluvium. Many of these soils supported grass, but some support hardwood or pine forests.

Usterts: These are the Vertisols in temperate areas that do not receive high amounts of rainfall during the summer. They have an ustic soil moisture regime. Cracks open and close once or twice during the year. They are found extensively in the USA, Australia, Africa, and India. If irrigated, Usterts can be used intensively, but large areas are used for grazing due to a lack of machinery to till soils.

Xererts: Xererts are the Vertisols of Mediterranean climates, which have xeric soil moisture regime. These soils have cracks that regularly close and open each year. In the USA, most of the soils supported grasses.

1.7 Soil Degradation

Land degradation is said to have taken place when the land within an ecosystem is no longer able to perform its environmental regulatory functions of accepting, storing, and recycling water, energy, and nutrients and when the potential productivity associated with a land-use system becomes non-sustainable (Oldeman et al. 1991). Again, soil degradation is considered as the measurable loss or reduction of the current or potential capability of soils to produce plant materials of desired quantity and quality. According to some authors (Blaikie and Brookfield 1987; Chisholm and Dumsday 1987; Blum and Eswaran 2004), land degradation is a broader term than soil degradation. But for synonymous use of the terms land and soil in most soil management literature, land degradation and soil degradation will be used interchangeably in the following sections.

Several physical, chemical, and biological processes are responsible for the degradation of soil (Lal 1994; Eswaran et al. 2001). The physical processes include deterioration of soil structure, crusting, hardsetting, compaction, erosion, and desertification. The chemical processes include leaching, fertility depletion, acidification, salinization, and pollution. The biological processes of soil degradation include reduction in carbon and decline in soil biodiversity. According to Beinroth et al. (1994), land degradation results from a mismatch between land quality and land use.

Yield reductions of 30–90 % by erosion in some root-restrictive shallow soils of West Africa were reported by Mbagwu et al. (1984) and Lal (1987). Erosion reduced 20–40 % yield of row crops in Ohio (Fahnestock et al. 1995) and elsewhere in Midwest USA (Schumacher et al. 1994). In the Andean region of Colombia, Ruppenthal (1995) have observed severe losses due to accelerated erosion on some lands. The productivity of some lands in Africa has declined by 50 % (Dregne 1990) as a result of soil erosion and desertification. Yield reduction in Africa due to past soil erosion may range from 2 to 40 %, with a mean loss of 8.2 % for the continent (Lal 1995). There are also serious productivity losses (20 %) due to erosion in Asia, including India, China, Iran, Israel, Jordan, Lebanon, Nepal, and Pakistan (Dregne 1992). Soil compaction has caused yield reductions of 25–50 % in some regions of Europe (Eriksson et al. 1974) and North America and between 40 and 90 % in West African countries (Charreau 1972; Kayombo and Lal 1994). In Ohio, reductions in crop yields are 25 % in maize, 20 % in soybeans, and 30 % in oats over a 7-year period (Lal 1996).

1.7.1 Causes of Soil Degradation

Soil degradation may result from natural causes and human-induced causes. Topographic and climatic factors such as steep slopes, frequent floods and tornadoes, storms and high-velocity wind, high-intensity rains, leaching in humid regions, and drought in dry regions are among the natural causes. Deforestation and overexploitation of vegetation, shifting cultivation, soil desurfacing, overgrazing, indiscriminate use of agrochemicals and lack of soil conservation practices, and overextraction of ground water are some anthropogenic causes of soil degradation.

The principal cause of land degradation is the non-appropriate land use. Economic and social problems, population pressure, poverty, land tenure system, farming systems, lack of technical advice, use of improper implements, etc. are the reasons of this mismanagement. GLASOD identified five main causes of human-induced soil degradation worldwide: deforestation, overgrazing, mismanagement of agricultural land, overexploitation, and bio-industrial activities. The causes of soil degradation are elaborated in the following sections.

1.7.1.1 Deforestation

Deforestation refers to the conversion of a forest into a non-forest use such as farmland, ranches, pasture, industrial complexes, and urban areas (Fig. 1.3). The world's total forest area is just over four billion hectares. Seven countries (Russia, Brazil, Canada, the USA, China, Indonesia, and the Democratic Republic of Congo) account for more than 60 % of the total forest area. Ten countries or areas have no forest at all, and an additional 54 countries have forest on less than 10 % of their total land area (FAO 2010). About one-half of the forests that covered the earth have been cleared (Kapos 2000). Each year, another 16 million hectares disappears. The World Resources Institute (1997) estimates that only about 22 % of the world's original forest cover remains intact – most of this is in three large areas: the Canadian and Alaskan boreal forest, the boreal forest of Russia, and the tropical forest of the northwestern Amazon Basin and the Guyana Shield (Guyana, Suriname, Venezuela, Columbia, etc.). For millennia, humankind has influenced the forests, and the

Fig. 1.3 Deforestation and soil degradation. One can hardly imagine that this was once a dense tropical forest (Photo courtesy of Mr. Dilip Kumar)

impact has been enormous. Deforestation is expanding and accelerating into the remaining areas of undisturbed forest, and the quality of the remaining forests is declining. Until quite recently, most of the deforestation occurred in Europe, North Africa, and the Middle East. By the beginning of this century, these regions had been mostly converted from the original cover. Now, deforestation in these regions has stabilized, and regrowth is occurring in some places, although the resulting secondary forests are of a different character. In the last few decades, the vast majority of deforestation has occurred in the tropics – and the pace still accelerates. The removal of tropical forests in Latin America is proceeding at about 2 % per year. In Africa, the pace is about 0.8 % per year and in Asia it is 2 % per year. According to the FAO analysis, deforestation was concentrated in the developing world, which lost nearly 200 million hectares between 1980 and 1995 (FAO 2000). Five tropical countries with the greatest total area of deforestation are Indonesia, Sudan, Myanmar, and the Democratic Republic of Congo. This loss is partially counterbalanced by reforestation efforts, new forest plantations, and the gradual regrowth and expansion of forested area in developed countries. However, the remaining natural forests are also highly degraded if not deforested. Measuring the extent and rate of deforestation is not as simple as it might at first appear (Singh et al. 2001). The first challenge is to define what is meant by a "forested area." In other words, what density of tree cover is required for an area to be considered a forest? The "continuous fields" tree cover map prepared by the Global Land Cover Facility (DeFries et al. 2000) shows that far from being homogenous, land areas can vary from 10 to 100 % forest cover and still be considered forests. Today, only 28 million km^2 is in so-called closed forests of 40 % canopy cover or above (Singh et al. 2001).

The FAO analysis concludes that the leading causes of deforestation are the extension of subsistence farming (more common in Africa and Asia) and government-backed conversion of forests to other land uses such as large-scale ranching (most common in Latin America and also Asia). Poverty, joblessness, and inequitable land distribution, which force many landless peasants to invade the forest for lack of other economic means, continue to drive forest clearance for subsistence farming in many regions. Often, people move into forest areas as logging activity creates roads that open formerly inaccessible regions. As for centrally planned forest conversion schemes, these are often used to spur short-term economic development, gain better political control of remote forest regions, and expand agricultural output (FAO 2000). Geist and Lambin (2001) have identified a host of causes for tropical deforestation. It is increasingly evident that a concentration of variables interacts across spatial and temporal scales (Turner et al. 2001). In Latin America, small farmer agricultural expansion along forest frontiers is probably the primary proximate cause of forest clearing, followed by in situ agriculture and pasture expansion, timber felling for fuel and construction, and infrastructure expansion. In Africa, a greater proportion of deforestation come from sedentary (non-frontier) agricultural expansion and fuel wood harvesting. In Asia, deforestation continued due to increasing demands on timber resources and continued migration of shifting farmers to remote areas.

Deforestation processes include industrial logging, clear felling, indiscriminate cutting, forest fires, shifting cultivation, and encroachment. The effects of deforestation are loss of forest, loss of biodiversity, climate change, and natural disasters such as cyclones, flood and drought, disruption of water cycle, decline in water quality, soil erosion, and sedimentation. Forests regulate the hydrologic processes: evaporation, transpiration, infiltration, and surface flow. Cutting trees in large tracts results in the climate getting drier in that area. Forest clearance exposes the bare soil to the scorching effect of the sun and the beating action of the rains. Significant amount of top soil is lost by erosion. Due to elevated temperature, soil organic matter is decomposed at a faster rate, and soil aggregates are broken down by raindrop impact. Infiltration rate is reduced and more water runs off. Large tracts of land become permanently impoverished due to soil erosion for these reasons. Deforestation increases the carbon dioxide emission into the atmosphere. Tropical deforestation accounts for about 20 % of total global carbon dioxide (CO_2) emissions (IPCC 2007).

Global Climate Models (GCM) may show that tropical deforestation would affect both small-and large-scale climate in the tropics (Zhang et al. 1996b). Results have further indicated that impacts of tropical deforestation depend upon regional climate characteristics in the tropics where disturbances are imposed (Mylne and Rowntree 1992; Polcher and Laval 1994). Tropical deforestation contributes to the increasing atmospheric carbon dioxide burden and hence the warming of climate (Houghton et al. 1996). Forests occupy an important position in the hydrological cycles of watersheds. Water evaporates from the soil and vegetation, condenses into clouds, and falls again as rain in a perpetual self-watering cycle. In addition to maintaining tropical rainfall, the evaporation cools the earth's surface. In many computer

Fig. 1.4 Preparing land for shifting cultivation (Photo courtesy of late German biologist Dr. Josef Margraf, with permission from his wife Mrs. Minguo Li-Margraf)

models of future climate, replacing tropical forests with a landscape of pasture and crops creates a drier, hotter climate in the tropics. Some models also predict that tropical deforestation will disrupt rainfall pattern far outside the tropics, including China, northern Mexico, and the south central USA. When people clear the forests, usually with fire, carbon stored in the wood returns to the atmosphere, enhancing the greenhouse effect and global warming. Once the forest is cleared for crop or grazing land, the soils can become a large source of carbon emissions, depending on how farmers and ranchers manage the land. In places such as Indonesia, the soils of swampy lowland forests are rich in partially decayed organic matter, known as peat. During extended droughts, such as during El Niño events, the forests and the peat become flammable, especially if they have been degraded by logging or accidental fire. When they burn, they release huge volumes of carbon dioxide and other greenhouse gases. When the ground surface is stripped of vegetation, the upper soils become vulnerable to erosion. Soil is washed into rivers and streams. This destroys the ability for the land to regenerate because of the loss in soil fertility. It also deteriorates water quality. In several parts of the world, entire sections of countries have been rendered unproductive because of soil erosion.

1.7.1.2 Shifting Cultivation

Shifting cultivation or slash-and-burn agriculture is an ancient farming system usually practiced by the indigenous people in the hilly areas of the humid tropics. In this system, a patch of forest, usually on gentle slopes or on summits of hills, is cleared, vegetation is slashed and burned (Fig. 1.4), holes are dug in soil with elementary tools, and seeds of assorted crops are sown before monsoon. Seeds germinate and grow rainfed, and the crops are harvested after 6–8 months. Then the land is left fallow. Farmers clear a new patch of forest for cultivation in the next season. Earlier, they returned to the previous land for cropping after 15–20 years, but the rotation period has now alarmingly squeezed to less than 3 years. Overpopulation, settlement of plain landers to hills, and scarcity of available land for cropping are behind this shrinkage. This has enhanced deforestation, loss of biodiversity, and soil degradation. It is estimated that each year approximately $1.9–3.6 \times 10^6$ ha land of primary close forests, $3.4–40 \times 10^6$ ha land of secondary close forests, and $6.9–21.9 \times 10^6$ ha land of secondary open forests are being lost due to shifting cultivation (Detwiler and Hall 1988).

Shifting farmers burn the slashed debris so that nutrients in its ash should increase fertility and enhance crop productivity. The effect of burning is, however, temporary. Rains following burning remove bases rapidly by erosion and leaching. Growing evidence suggests that shifting cultivation in its distorted form leads to an adverse effect on soil and water. Such activities affect soil physical and chemical properties, reduce nutrient stocks, and accelerate soil erosion and sedimentation. Gafur et al. (2000) estimated soil loss of 43 Mg ha^{-1} $year^{-1}$ due to shifting cultivation in Bandarban Hills, Bangladesh.

1.7.1.3 Overgrazing

According to a UK regulation, overgrazing means grazing land with livestock in such numbers as to adversely affect the growth, quality, or species composition of vegetation on that land to a significant degree (Statutory Instrument 1996). Wilson and MacLoad (1991) include animal performance as well; they state that a grassland is overgrazed where a concomitant vegetation change and loss of animal productivity arises from herbivores' grazing of land. Overgrazing can mean different things to the grazier and the range manager. For the grazier, it implies that the pasture can no longer carry as many animals as before, or that its productivity has declined. Overgrazing is a major cause of soil degradation worldwide (Oldeman et al. 1991), accounting for 35.8 % of all forms of degradation. However, degradation caused by overgrazing is especially widespread in Australia and Africa, where it accounts for 80.6 % and 49.2 %, respectively, of all soil degradation and least extensive in Europe (22.7 %) (Warren and Khogali 1992).

Overgrazing is the most devastating cause of desertification in arid lands. Livestock are the main source of income in many arid and semiarid countries. Domestic stocks are widely diverse and consist of camels, donkeys, horses, cows, sheep, and goat. The forage and overgrazing of livestock cause a chain of degradation, critically reducing vegetation cover and soil fertility, as well as increasing erosion. Domestic animals rapidly clear vegetation, placing stress on a land that already has a low vegetation cover (Fig. 1.5). They also move in large groups and have sharp hooves that easily break up the soil, leaving it susceptible to erosion. Erosion decreases fertile organic content of the soil. The lack of organic matter can lead to desertification through reduced nutrient availability for plant growth.

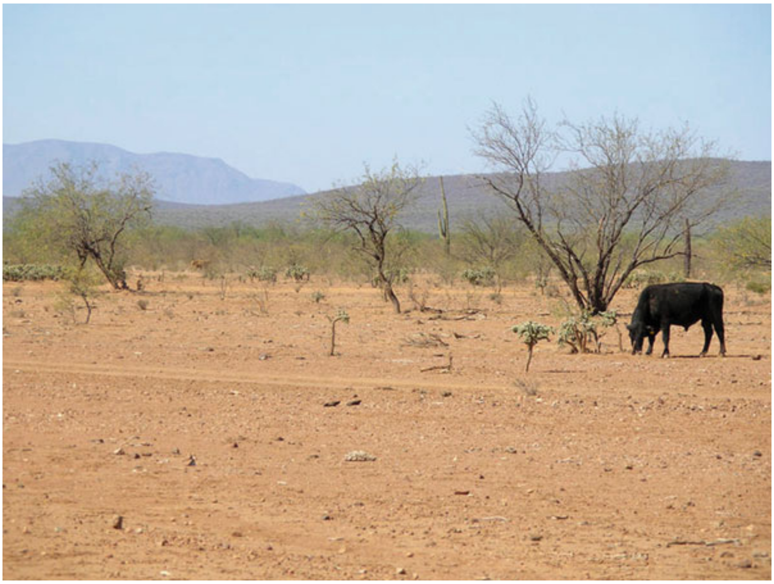

Fig. 1.5 The soil has badly suffered from overgrazing (Photo courtesy of Sky Jacobs of Wild Sonora)

Abril and Bucher (1999) measured the changes in soil characteristics, nutrient availability, and microbial activity on sites utilized by different grazing intensities in Argentina. Three sites were selected for comparison: a highly restored (no grazing for 20 years), a moderately restored (8 years of restoration), and a highly degraded (extremely overgrazed). The following parameters decreased as the grazing intensity increased: the soil moisture (4.5–2.25 %), the organic matter (4.68–1.45 %), and the nitrogen content (0.28–0.14 %). Microbial activity ranged from 0.89 at the restored sites to 0.22 mg CO_2/g/week at the highly degraded site. The seasonal variations in the density and the activity of microorganisms increased from the highly restored to the highly degraded site, probably as a response to an increased lack of humidity. The cellulolytic and nitrifier groups were the most affected, whereas the ammonifier and free-living N-fixing organisms decreased in the highly degraded site only. N fixation was more intense at the moderately restored site followed by the highly degraded site. The observed values are interpreted as resulting from the interaction between organic matter availability (as energy source) and N deficiency. The results suggest a strong influence of overgrazing on the soil fertility, as well as on the soil ability to buffer water stress during the dry season. According to Ling Hao et al. (1997), an average of 12.4 % of the total carbon initially stored in soils (0–20 cm soil layer) has been lost due to overgrazing over the 40-year period.

Fig. 1.6 Brickfields in agricultural land (Photo courtesy of Dr. Animesh Biswas)

1.7.1.4 Soil Desurfacing

Soil desurfacing refers to the removal of a layer of surface soil for many different purposes, including construction of road and railroads and making of bricks. Brick is an important element of urban development. Most brickfields are located in fertile alluvial lands for the presence of soils of suitable textures. For example, hundreds of brickfields are seen in agricultural lands along both sides of the river Buriganga near Dhaka, Bangladesh (Fig. 1.6). A more or less a uniform layer of soil, 15–20 cm thick, is collected from the surface of huge areas of paddy fields. An estimate shows that a five-story building of 250 m^2 floor space needs 500,000 bricks. More than 1 ha furrow slice surface soil (100 m × 100 m × 15 cm) would be needed to make this number of bricks. The surface soil is the most fertile part of the soil; it contains the highest organic matter and nutrients and possesses the most suitable physical and chemical conditions for plant growth. Soil desurfacing decreases organic matter, nutrients, and available water, and the exposed subsoil is more compact. Soil desurfacing significantly reduces soil quality and crop yields (Grewal and Kuhad 2002).

1.7.1.5 Monocropping

Monocropping is the practice of growing a single crop year after year on the same land, in the absence rotation through other crops. Rice, maize, soybean, and wheat are the common crops grown as monocrops. It is economically an efficient system, but it can negatively impact the soil fertility and ecology. The roots of the crop draw the same kind and proportion of nutrients from the soil for a long period of time. The nutrient status of the soil becomes imbalanced after a prolonged period of cropping. Particular types of insects and pests proliferate. Farmers become increasingly dependent on pesticides.

1.7.1.6 Mismanagement of Irrigation

It is not possible for farming to occur without artificial irrigation in arid lands where there is a deficiency of moisture. However, irrigation in arid lands can further enhance desertification through salinization and alkalinization. Salinization occurs when irrigation water evaporates quickly, leaving natural salts (chlorides, sulfates, and carbonates of sodium, potassium, calcium, and magnesium) at the surface of the soil. Over a long time, excessive quantities of salts accumulate at or near the soil surface, making it increasingly difficult for plants to extract water from the soil. This is because inadequate water is applied for the scarcity of water in the arid and semiarid climate. Irrigation water must consider the leaching requirement (extra water to leach the salts) of the soil.

1.7.1.7 Use of Heavy Farm Machineries

Heavy farm machineries are used in mechanized agriculture by the developed countries. Tractors and cultivators may weigh more than 20 tons. Such heavy loads of these machines severely compact soils. This is almost irreversible. Deep compaction below the plow layer (plow pan) is difficult to decompact. Impact of heavy farm machineries on soil degradation is discussed in more detail in connection with soil compaction (Chap. 2; Sect. 2.2.3.1).

1.7.1.8 Mining

Mining contributes significantly to soil pollution, especially with heavy metals. The disposal of Cd from Zn mines in the water of Jintzu River in Japan and contamination of soils and rice grains that caused the notorious itai-itai disease is a good example. A soil survey conducted by Kurnia et al. (2000) in West Java exhibited that the soil surrounding the traditional gold mining areas was polluted by mercury. A high concentration of mercury was found in rice straw and rice grain in that area. All of the values were higher than the maximum permitted level of Hg in soils (0.5 mg kg^{-1}). Mining is responsible for large-scale soil degradation.

1.7.1.9 War and Ammunitions

Weapons and ammunitions are a source of soil pollution in regions of conflict. Firing ranges often leave soils contaminated with metals from spent bullets. Lead is the primary soil contaminant of concern in these ranges. The normal operation of a range can produce lead concentrations of several percent (1 % equals to 10,000 mg kg^{-1}; safe level is only 35 mg kg^{-1}) in soils adjacent to targets. Soil samples from a weapon destruction facility in Belgium showed high concentration of arsenic, copper, and lead. Depleted uranium is an important source of soil pollution. It is highly toxic and has been used in active combat by the US and UK forces. This material has as much as 75 % of the strength of enriched uranium; so it brings lasting health problems with it wherever it is used. Two hundred eighty-six metric tons of depleted uranium was used during the first Gulf War in 1991. Defoliating agents were used by the American Army in the Vietnam War. These defoliants continue to cause death and environmental problems today. Agent Orange contained dioxin which is one of the most potent chemicals. In Vietnam there are areas where the levels of dioxins in the soil exceed internationally accepted levels by as much as one hundred times.

1.7.1.10 Indiscriminate Waste Disposal

Materials that have reached the end of their useful life are called wastes (Hill 2010). Wastes are of various kinds: industrial, municipal, agricultural, domestic, and nuclear. These wastes are the chief sources of soil pollution (Alloway 1995). Municipal wastes include domestic and kitchen wastes, market wastes, hospital wastes, livestock and poultry wastes, slaughterhouse wastes, and metals, glass, and ceramic wastes. Municipal wastes can be biodegradable or nondegradable, recyclable materials, or inert. Municipal wastes may contain nonbiodegradable organics and heavy metals. Sewage sludge is a municipal waste often used as composts in crop fields because it contains considerable amounts of plant nutrients. But for heavy metal (Pb, Cd, Zn, etc.) contamination of soil, the use of sewage sludge as a fertilizer is discouraged. Hospital wastes include general wastes, sharps, non-sharps, blood, body parts, chemicals, pharmaceuticals, medical devices, and radioactive materials. Potentially infectious waste includes all waste items that are contaminated with or suspected of being contaminated with body fluids. These wastes are often disposed with municipal wastes in open dumps and on lands (Fig. 1.7).

1.7.1.11 Use of Agrochemicals

Agrochemicals include fertilizers, manures, soil conditioners, hormones, and pesticides. Fertilizers and manures introduce some heavy metals such as arsenic, cadmium, uranium, and vanadium. Phosphate fertilizers contain considerable amount

Fig. 1.7 Open dumping of municipal wastes at the edge of Chittagong Metropolitan area, Bangladesh (Photo courtesy of Dr. Animesh Biswas)

of Cd. Poultry and pig manures add some zinc, arsenic, and copper. Pesticides, another group of agrochemicals, include insecticides, herbicides, and fungicides. Some organophosphorus compounds, organochlorines, the carbamates, and the pyrethroids are used as pesticides. They are highly toxic chemicals, and some are persistent. Their indiscriminate use is not desirable. Soil degradation due to agrochemicals has been described in soil pollution (Chap. 6).

1.7.2 Institutional Initiatives for Assessment of Land/Soil Degradation

Current status of soils is not widely known on a global scale, although wide information on some other natural resources like forests, water, air, and biodiversity may be obtained. The FAO–UNESCO Soil Map of the World is the only available map, which has a fully global coverage. This map is at a scale of 1:5M, based on soil survey data prior to the 1970s. The United Nations Environment Programme (UNEP) formulated a project proposal for Global Assessment of the Status of Human-Induced Soil Degradation (GLASOD) in 1987 which developed methodologies to create soil and terrain databases. The main objective of the GLASOD project was to strengthen global awareness of policy makers and decision makers of the dangers resulting from inappropriate land and soil management. The project also aimed to provide information (in the form of digitized maps) on soil and terrain resources to the potential users in nontechnical terms. Since then (1991), important regional inventories on land degradation were undertaken, notably in South and Southeast Asia in close cooperation with national institutes in 17 Asian countries and with technical and financial support from UNEP and FAO. In these surveys the GLASOD methodology has been refined. The GLASOD map does not, however, assess the vulnerability of land resources to the soil degradation process but the status of degradation which assists policy makers and resource managers to identify priority areas of intervention. The SOTER (Soil and Terrain Database) project was initiated by the International Society of Soil Science in 1986; SOTER was developed as an internationally endorsed land resource information system that can store at different levels, detailed soil and terrain attributes in such a way that these data can be assessed, combined, and updated immediately and can be easily analyzed from the point of view of potential land use, in relation to food requirements, environmental impact, and conservation (van Lynden 2004). The LADA (Land Degradation Assessment for Dryland Areas) project aims to develop and test an effective methodology to assess causes, status, and impact of land degradation in dry lands in order to improve decision making for sustainable development in dry lands at local, national, subregional, and global levels. LADA is an international UN initiative, supported by the Global Environment Facility (GEF), UNEP, UNCCD, and FAO and is being executed by FAO. The Global Assessment of Land Degradation and Improvement (GLADA) is intended to identify (1) the status and trends of land degradation and (2) hotspots suffering extreme constraints or at severe risk, and, also, areas where degradation has been arrested or reversed. Among other ongoing activities to assess the status of soils in the world is the World Overview of Conservation Approaches and Technologies (WOCAT). WOCAT is documenting and evaluating soil and water conservation case studies worldwide. WOCAT has developed a new internationally recognized standardized methodology, involving a set of three comprehensive questionnaires, for documentation and evaluation of individual soil water conservation technologies and approaches, and including area coverage. The latter also includes assessment of soil degradation following a method similar to GLASOD and ASSOD (Bai et al. 2008, 2010).

1.7.3 Types of Soil Degradation

A world map on the status of human-induced soil degradation was published by ISRIC, in cooperation with FAO and UNEP in 1991. In preparation of the map, a general classification was developed, referred to as the GLASOD (Global Assessment of Soil Deterioration) classification. According to GLASOD, there are five main types of soil degradation, including water erosion, wind erosion, chemical deterioration, physical deterioration, and degradation of biological activity (Oldeman 1991). There are several subtypes of each type except biological degradation. These types and subtypes are mentioned below.

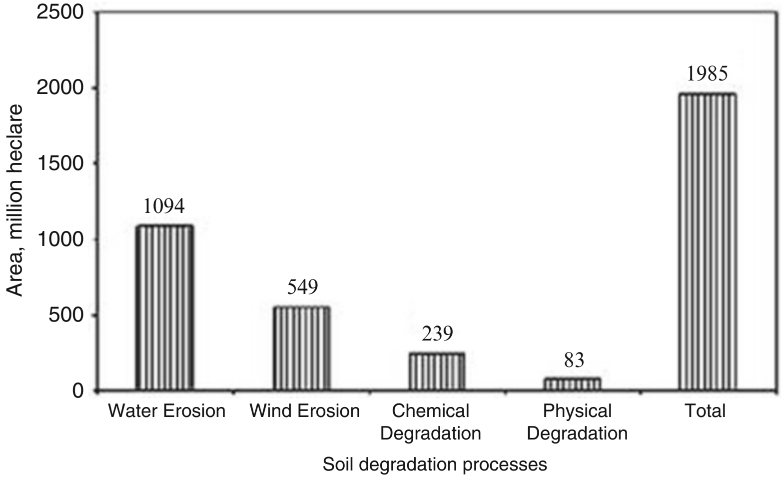

Fig. 1.8 Land area so far degraded by different processes (Data from Oldeman 1994)

Soil degradation types and subtypes

Type	Subtypes
W: Water erosion	Wt: loss topsoil Wd: terrain deformation/mass movement Wo: off-site effects Wo: reservoir sedimentation Wof: flooding Woc: coral reef and seaweed destruction
E: Wind erosion	Et: loss of topsoil Ed: terrain deformation Eo: overblowing
C: Chemical deterioration	Cn: Loss of nutrients and/or organic matters Cs: Salination Ca: Acidification Cp: Pollution Ct: Acid sulphate soils Ce: Eutrification
P: Physical deterioration	Pc: compaction, sealing, and crusting Pw: water logging Pa: lowering of water table Ps: subsidence of organic soils Po: other physical activities such as mining and urbanization
B: Degradation of biological activity	

Land area degraded by each type of soil degradation is shown in Fig. 1.8.

Table 1.3 Extent of degraded lands in different regions of the world

	Agricultural land			Permanent pasture			Forests		
Region	Total Mha	Degraded Mha	%	Total Mha	Degraded Mha	%	Total Mha	Degraded Mha	%
Africa	187	121	65	793	243	31	683	130	19
Asia	536	206	38	978	197	20	1,273	344	27
South America	142	64	45	478	68	14	896	112	13
Central America	38	28	74	94	10	11	66	25	38
North America	236	63	26	274	29	11	621	4	1
Europe	287	72	25	156	54	35	353	92	26
Oceania	49	8	16	439	4	19	156	12	8
World	1,475	562	38	3,212	685	21	4,041	719	18

Kertesz (2009), FAO (1990), and Scherr (1999)

1.7.4 Extent of Soil Degradation

Some 38 % of the agricultural area of the earth can be considered as degraded. The share of degraded territories in Africa is 65 %, in Central America 74 %, and in South America 45 %. The proportion of degraded pasture and forests is much smaller 21 % and 18 %, respectively. Considering only used land (agricultural area, permanent pasture, and forests), the proportion of degraded area is 23 % and that of strongly degraded land is 14 %. Table 1.3 shows the distribution of degraded lands in different regions of the world.

An estimated 38 % of the area (749 M ha) affected by human-induced soil degradation was slightly degraded, 46 % (910 M ha) was moderately degraded, 15 % (296 M ha) was strongly degraded, while less than 1 % (9.3 M ha) was extremely degraded (Oldeman 1994). Adverse effects on soil health and soil quality in Asia arise from nutrient imbalance in soil, excessive fertilization, soil pollution, and soil loss processes (Zhang et al. 1996a, b; Hedlund et al. 2003). In Africa, three-quarters of farmland is severely degraded (Eswaran et al. 1997; Stocking 2003). As a result, Africa cannot produce enough food to keep pace with its needs, and per capita food production is declining largely due to loss of soil health and soil quality (Lal et al. 1997; Lal 1998).

1.8 Laws of Sustainable Soil Management

Soil resources can never be taken for granted. Many civilizations, including Mayan, Incas, Indus, and Mesopotamia, are now extinct because they ignored their soil resources. In the context of increased global threat of soil degradation, Lal (2009) proposed ten "laws of sustainable soil management." The laws are summarized below:

(1) Soil resources are unequally distributed among biomes and geographic regions. Highly productive soils are located in favorable climates and are highly populated and are converted to managed ecosystems, such as croplands, pasture, forest, and energy plantations. (2) Most soils are prone to degradation by land misuse and soil mismanagement. Desperate actions of resource-poor farmers, smaller landholders, and greedy, shortsighted lust for quick economic returns by humans lead to soil degradation. (3) The way how the soil is managed affects soil erosion and decline in soil quality by other degradation processes. Restorative and recommended soil and water management practices may retain productive potential of farming systems. Judicious management of on-site and off-site inputs is needed for sustainable use of soil. Indiscriminate and excessive use of tillage, irrigation, and fertilizers can aggravate soil degradation. (4) The rate and susceptibility of soil to degradation increase with increase in mean annual temperature and decrease in mean annual precipitation. All other factors remaining the same, soils in hot and arid climates are more prone to degradation and desertification than those in cool and humid ecoregions. (5) Soil can be a source or sink of greenhouse gases, CO_2, CH_4, and N_2O, depending on land use and management. Soils are a source of radiatively active gases with extractive farming which create a negative nutrient budget and degrade soil quality and a sink with restorative land use and judicious management practices which create positive C and nutrient budgets and conserve soil and water while improving soil structure and tilth. (6) Soils are nonrenewable over a human time frame scales, but are renewable on a geological scale. (7) Soil's resilience to natural and anthropogenic perturbations depends on its physical, chemical, and biological processes. Favorable chemical and biological processes enhance resilience only under optimal soil physical properties and processes, including structure, tilth, aeration, water retention and transmission, and temperature. (8) The rate of restoration of the soil organic matter pool is extremely slow, while that of its depletion is often very rapid. In general, restoration occurs on a centennial time scale and depletion on a decadal time scale. (9) Soil structure depends on stability and continuity of macro-, meso-, and micropores which are the sites of physical, chemical, and biological processes that support soil's life support functions. Sustainable management systems enhance stability and continuity of pores and voids over time and under diverse land uses. (10) Sustainable management of agricultural ecosystems implies an increasing trend in net primary productivity per unit input of off-farm resources along with improvement in soil quality and ancillary ecosystem services such as increase in the ecosystem C pool, improvement in quality and quantity of renewable fresh water resources, and increase in biodiversity.

Study Questions

1. Explain that soil is a limited natural resource. It is a renewable natural resource in geological time frame, but it is a nonrenewable natural resource in human life time frame.

2. What do you mean by land capability? Discuss the management requirements for land capability Classes I, II, III, and IV. Why should lands of capability Classes VII and VIII be left under their natural state?
3. Define soil fertility, soil heath, soil productivity, soil resilience, soil performance, and soil quality. Explore relationships among them. Compare characteristics of soil quality of Classes I and IX.
4. What are the causes of soil degradation? What are the types of soil degradation? Discuss the extent of global soil degradation.
5. Discuss the institutional initiatives for the assessment of global soil degradation. Mention the laws of sustainable soil management.

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Chapter 2
Physical Deterioration of Soil

Physical deterioration of soil involves the destruction of soil structure, dispersion of soil particles, sealing of pores, compression and increasing density, consolidation, compaction and reduced root penetration, low infiltration, waterlogging and runoff, and accelerated erosion. Along with denudation, these processes lead to desertification in the arid and semiarid regions. The ultimate result is the loss of the capacity to support ecosystems. There are about 68.3 M ha compacted soil globally which accounts for 4 % of human-induced soil degradation. Soil compaction is a global problem associated with mechanized agriculture. It has caused yield reductions of 25–50 % in some regions of Europe and North America and between 40 and 90 % in West African countries. Planned and integrated management involving reduced, conservation and timely tillage, generous addition of organic matter, mulching, crop rotation, subsoiling, and controlled trafficking are recommended for sustainable use of compacted soils. Desertification is a process of land degradation occurring mainly in the arid and semiarid regions due to human actions on ecosystems combined with adverse climatic conditions. The United Nations estimates that desertification affects 70 % of arid lands, amounting to about 30 % of the world cultivable land.

2.1 Processes and Types of Physical Deterioration of Soil

Important physical properties of soils related to soil fertility, productivity, and soil quality are texture, structure, bulk density, porosity, soil water, air, and temperature. Of these properties, only soil texture is a permanent property that cannot be easily altered. The other properties may be readily changed by soil management practices. Soil structure may be modified by tillage and addition of manures. Tillage operations break the clods and peds and modify the size of pores and the proportion of large and small pores. Consequently, the bulk density, water holding capacity, and aeration status are modified. Often, tillage operations destroy soil structure, reduce porosity, and make the soil compact. Thus, physical deterioration of the soil may

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take place due to faulty soil management. It has been observed that the following processes may account largely for the physical deterioration of soil.

1. *Coalescence*: Slow deposit of fine soil particles between individual aggregates, welding them together into a massive structure. Soils that have been tilled in a very dry state are particularly prone to coalescence.
2. *Slaking*: Structural collapse when dry soil aggregates are wetted rapidly by rain or irrigation. Soils with low organic carbon (<2 %) are subject to rapid slaking.
3. *Dispersion*: Disintegration of the slaking products into individual particles due to high forces of repulsion between clay particles. These high forces are promoted by high concentrations of exchangeable sodium.
4. *Consolidation*: Reduction of soil volume by destruction of pore space, mainly large pores. Soils with low organic carbon (<2 %) are particularly prone to compaction, but soils high in organic matter exhibit elastic resilience that makes them rebound.
5. *Aggregate Pulverization*: This happens when soils that are too dry are tilled, which destroys aggregates into fine powder. On the other hand, tilling when soil is too wet destroys aggregates by smearing, because soil strength is at a minimum. Tillage should ideally happen when soil moisture is at a point called "lower plastic limit."

The physical deterioration is one of the major types of soil degradation. It has the following subtypes: (1) surface sealing, surface crusting, hardsetting, and compaction (Pc); (2) waterlogging (Pw); (3) lowering of water table (Pa); and (4) subsidence of organic soils (Ps). Desertification is also a physical degradation process which has become a major concern in the arid and semiarid regions.

2.2 Surface Sealing, Crusting, Hardsetting, and Compaction (Pc)

2.2.1 Surface Sealing

The term soil sealing is used to describe a change in the nature of the soil (usually surface soil) leading to impermeability. Soil particles are dispersed by the collapse of soil aggregates due to physical pressures, such as raindrop impact, or due to a chemical agent such as excess exchangeable sodium. Structure deterioration can also take place for improper tillage operations. For example, peds are ground to fine powders when soils are tilled in the dry condition. Fine particles enter into the macropores and reduce infiltration and percolation. On the other hand, tilling in the wet condition puddles the soil. Unstable peds may slack when wetted. Thus, the surface soil becomes impervious to water and plant roots. Sealed soils are lost to many uses such as agriculture and forestry while the ecological soil functions are severely impaired or even prevented. In addition, surrounding soils may be influenced by change in water flow patterns or the fragmentation of habitats. Current studies suggest that soil sealing is nearly irreversible.

Soil hydrology may be severely impacted by surface sealing. For example, a surface seal will greatly reduce water movement into soil (Ahuja 1983). Seal formation increases soil density and reduces porosity, pore size, and pore continuity. McIntyre (1958) found that a seal consist of two parts: (1) an upper skin zone about 0.1 mm attributed to compaction by raindrop impact and (2) a washed-in zone of about 1.5 mm of decreased porosity attributed to the movement of particles into the soil with water. Agassi et al. (1981) suggested that there are two complementary mechanisms of a seal formation: (1) a physical breakdown of soil aggregates, caused by wetting and raindrop impact, and (2) physicochemical dispersion of clay particles, which move into the soil with the infiltrating water, that clog pores to form a washed-in layer of low permeability. Decreased total porosity increases bulk density (Assouline 2006; Eynard et al. 2004) and slows solute transport (Assouline 2006; Huang and Bradford 1993) and root growth (Lynch and Bragg 1985). Macropores (pore diameters >1,000 μm) increase pathways for water that often increase infiltration and reduce runoff. But surface sealing reduces the number of macropores.

In agricultural soils, prolonged and repeated cultivation of surface soils during cropping regimes destroys natural aggregation. Soil aggregates may collapse due to plow pressure and by the load of heavy farm machineries and hoof-strokes of farm or grazing animals. As natural aggregates are destroyed, finer particles are released and, when exposed, disperse under both raindrop and irrigation action. Dispersed particles reorientate and fill in larger pores. The soils most sensitive to aggregate deterioration tend to be sandy loams, sandy clay loams, and sodic soils in the dry climates and coastal regions. Sodic soils susceptible to slaking and dispersion are particularly at risk following cultivation. Low organic matter containing soils cannot develop stable peds that could resist slacking.

In a study, Heil et al. (1997) reported that many Sahelian Alfisols are prone to sealing because of low soil organic matter content and exposure of finer-textured subsoil attributable to erosion. Their study sites were located on six soil series of the Hamdallaye watershed (500 ha), with soil textures ranging from sandy loam to sand, classified as Psammentic Kandiustalf and Petroferric Kanhaplustult. All seals sampled in the watershed were structural seals and were morphologically similar, with a 0.1–1.0 mm thick continuous plasmic clay layer within 4 mm of the surface. Organic C contents of sealed sites were very low (0.1–0.2 %) at 0–50 mm depth and slightly higher at unsealed sites. Aggregation was too weak to withstand raindrop impact. Extractable Fe and Al contents of the six soil series were related to clay content, which was likely the controlling factor of seal formation. With simulated rainfall of 90 mm h^{-1}, the same six soils formed a seal during the initial 30-min rainfall event in most cases, with no change in layer thickness thereafter. Soil with more than 15 % (silt + clay) content formed a 2-layer structural seal, whereas coarser textured soils developed 4-layer structural seals. The physical processes of soil slaking and sealing are the result of the kinetic impact of raindrops on the soil surface and the translocation of soil particles by flowing water. When the drop impact forces exceed the internal cohesion of the impacted soil aggregates, they break down into primary mineral particles. These particles are transported by surface runoff or washed into the soil surface layer (Schmidt 2010). When deposited the translocated particles could clog soil pores and form superficial layers characterized by higher bulk density and lower saturated hydraulic conductivity than the soil beneath (Betzalel et al. 1995). Due to the loss of soil water storage and infiltration capacities, soil erosion and the risk of flooding are substantially increased.

2.2.2 Surface Crusting

On further consolidation and drying, surface sealing gives rise to surface crusts. Sealing formation and crust are very common phenomena in many soils worldwide, especially in arid and semiarid soils. Rainfall causes a series of interactions between water and soils: disintegration, detachment, entrainment, deposition, and compression. These actions result in the formation of seal and, subsequently, the crust of soils. Crust is a thin layer at the soil surface characterized by a greater density, higher shear strength, and lower hydraulic conductivity than the underlying soil (Zejun et al. 2002). The mechanism of crust formation involves two main complementary processes: (1) physical action including disintegration of soil aggregates and soil particles compaction caused by impaction of raindrop and (2) physical-chemical action including dispersion of aggregates, movement of soil particles that clog the conducting pores and form a less permeable layer at topsoil region (Cai et al. 1998). The formation of seal and crust depends on many factors, including the texture and stability of soil, intensity and energy of rainfall, gradients and length of slope, and electrolyte concentration of the soil solution and rainwater (Remley and Bradford 1989).

There are two types of surface crusts: structural crust and depositional crust. A structural crust is a surface layer of the soil, a few millimeters to a few centimeters thick, more compact than the material beneath. The import of external materials is not involved in the formation of the structural crust. Structural crusts are developed also due to trampling by livestock or through traffic by agricultural machinery. Structural crusts may be hardsetting crusts and traffic crusts. Hardsetting is a process of compaction of soil along with increased bulk density occurring without the application of an external load. The term hardsetting was first used by Northcote (1960) in his soil classification system of Australia. Hardsetting is a characteristic of soil horizons, usually cultivated seedbeds, which contain unstable soil aggregates. Soil aggregates collapse and the seedbed slumps when the soil is wet, and a hard, structureless mass of soil results upon drying. Hardsetting, however, involves a much greater thickness of material, which commonly includes not only the A1 or Ap horizon but also the E horizon (Greene 2005).

Hardsetting is also a surface crust which involves the collapse of some or all of the aggregates during and after wetting of previously loosened topsoil. The hardsetting processes can be divided into two physically distinct processes: slumping and uniaxial shrinkage. Slumping occurs during and after the wetting of a soil containing water-unstable aggregates. The aggregates soften and swell simultaneously, and some or all of the finer particles (silt and clay) become suspended. Some of the clay fractions disperse. Aggregates disintegrate because they have insufficient strength to withstand the stresses set up by rapid water uptake, caused by rapid release of heat on wetting, trapped air, the mechanical action of rapidly moving water (Collis-George and Greene 1979), or by differential swelling. Uniaxial shrinkage is important because the closer proximity of particles increase the strength upon drying hardsetting soils. Laboratory experiments on the behavior of aggregate beds of a hardsetting soil, wetted under tension or at zero potential, show that at least during the early stages of drying, uniaxial shrinkage occurs. Mullins et al. (1987) have proposed the following explanation for the development and increase in strength observed in hardsetting soils, starting with a cultivated bed consisting of dry aggregates: wetting of the system mobilizes some or all of the silt and clay. This may occur through slaking and/or dispersion. During the early stages of drying, the mobilized material is carried behind the retreating water meniscus to occupy concavities on the surface of sand grains and any remaining aggregates, forming annular bridges between them (Kemper et al. 1987). Traffic crusts are formed by the external pressure of farm machinery and animals which can cause a serious reduction in penetrating water and seedling emergence. Overgrazing can induce crust formation by two mechanisms: surface compaction of wet or moist soils and mechanical destruction of the surface soil aggregates. Some soils are naturally and genetically hardsetting. They are abundant in tropical areas (Fabiola et al. 2003). Naturally, hardsetting soils are unable to develop water-stable aggregates. Hardsetting condition can occur in soils with high exchangeable sodium percentage. Some hardsetting soils are impervious, compacted as well as cemented. Amorphous silica and imogolite-like aluminosilicates may act as cementing agents (Chartres et al. 1989).

On the other hand, a depositional crust develops when soil particles, suspended in water, are deposited on the soil surface as the water infiltrates or evaporates. Externally derived materials are always involved in the construction of depositional crusts. Depositional crusts develop by deposition of suspended particles on the soil surface. These are found in some cultivated and noncultivated soils. The main sources of fine eroded soil particles are flood and furrow irrigation water, raindrop impact splash of loose soil particles, overflow and floods from rivers and runoff, and sheet erosion. The clay and silt particles in turbid suspension can either disperse or flocculate. They flocculate when the electrolyte concentration in the suspension exceeds the flocculation threshold of the clays (Oster et al. 1980). Depositional crusts formed from flocculated particles have an open structure and high permeability. Conversely, when the suspension electrolyte concentration is below the flocculation threshold, dispersed particles settle to form the depositional crust, the hydraulic conductivity of which is several orders of magnitude lower than that of the parent soil.

Sealing formation and crust can significantly reduce the infiltration of soil and increase runoff at the surface of soil, which in turn increase the transport capacity for entraining detached materials from soils (Levy et al. 1994). Surface crust and soil erosion are closely related that many scientists have paid attention to the formation of sealing and crust. Both soil crust and soil erosion involve particles' detachment and transport processes. Although most erosion models do not take account for the affection of crust to erosion, new concepts have been introduced which involve the explicit delineation between detachment and transport processes (Bissonnris et al. 1998). So it is believed that the new soil erosion model should take the basic concept of sealing and crust as one of its supporting theories.

2.2.2.1 Soil Factors Affecting Crust Formation

Crusting primarily involves the destruction of soil aggregates. Aggregates are mainly held together by electrochemical forces binding the clay mineral particles together. Aggregate stability depends on clay mineral types, their electrochemical characteristics and the electrolyte concentration of the soil solution. The more stable the aggregate of a soil is, the less susceptible it is to crusting.

Clay Minerals

Kaolinitic soils have usually stable aggregates because of edge-to-edge and edge-to-face connections between the clusters, since some positive charges exist on the broken edges, while the broad particle faces expose a negative charge. Kaolinite is a non-expanding or non-swelling type of clay. So, soil aggregates made up of dominantly kaolinite clay do not slake easily upon wetting. On the other hand, montmorillonite is an expanding clay that swells extraordinarily when wetted. Therefore, soils containing high proportion of montmorillonite form water-unstable aggregates. Such soils, for example, Vertisols, slake easily when wetted. They become sticky when wet and very hard when dry.

Exchangeable Cations

The formation of all soil crusts involves aggregate disintegration and dispersion. The dispersion of soil colloids is affected by nature and distribution of exchangeable cations held by electrical charges on colloidal surfaces. Exchangeable sodium (Na^+) is a much more dispersive cation than hydrogen (H^+), calcium (Ca^{++}), magnesium (Mg^{++}), and other polyvalent cations. As the percent of exchangeable Na^+ rises, a solution of much higher concentration is needed to cause flocculation. In other words, clay with a high exchangeable sodium percentage will disperse quickly under the dilution effect of rainwater at the soil surface.

Organic Matter

Organic substances may stabilize soil structure in two different ways: (1) organic substances can reduce interaction of water with inorganic colloids, and (2) it can bind soil particles together physically or chemically. Humic materials act as cements that bind particles together. The organic polymers expose along their long complex surfaces negative and positive edges of amides, carboxyl, hydroxyl, and other polarizable groups.

Iron and pH

Iron oxides may bind soil particles together. Kaolinitic soils rich in iron oxides and lower in pH may develop crusts.

2.2.2.2 Management of Soil Crust

Reduced infiltration is the most damaging problem of crusted soils. Low infiltration increases runoff and erosion and decreases available soil moisture for plants. There are three methods of augmenting infiltration: the use of soil amendments, conservation farming, and tillage management. These methods may be used separately or combined.

Soil Amendments

There are some soil amendments to prevent slaking and dispersion of soil aggregates. One way to reduce the risk of crusting is to improve soil structure and aggregate stability at the soil surface. Increasing electrolyte concentration by spreading phosphogypsum on the soil surface may be employed (Agassi et al. 1981). Phosphogypsum dissolves quite readily during rainstorms and releases Ca^{++} and SO_4^{--} ions into the soil solution to support concentrations high enough to prevent clay dispersion. Adding phosphogypsum improves the physical properties of the soils by replacing Na^+ by Ca^{++} on the soil colloids. Improving soil structure and reducing crust formation by using organic polymers (e.g., PAM–polyacrylamide) have been under intensive investigation for many years.

Conservation Farming

Conservation farming systems effectively reduce soil crusting. Conservation farming includes some specific tillage systems, such as no tillage, mulch farming, stale mulch, and minimum tillage. These systems are widely used in the USA and Australia to counteract water and wind erosion. There are two main goals for conservation farming: (1) to improve the physical and chemical conditions of the soil and (2) to shield the soil surface from the destructive force of the raindrops using mulches or protective cover. Integration of these two approaches commonly produces encouraging results. Cover crops and mulch reduce water loss and erosion appreciably.

Tillage Management

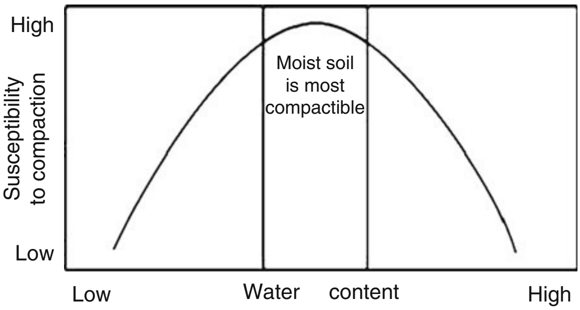
Tillage management for crusted soils includes choice of tillage implements, axle load, tillage operations such as number of turns of plow, and timing of tillage operations in relation to wetting and drying.

2.2.3 Soil Compaction

Soil compaction is the physical consolidation of the soil by an applied force, usually by tillage implements, that destroys soil structure, compresses soil volume, increases bulk density, reduces porosity, and limits movement of water and air within the soil. Surface sealing, crusting, and hardsetting are actually some sorts of soil compaction occurring in the surface soil by slaking of soil structure. Sometimes soil consolidation is distinguished from soil compaction. Soil consolidation is regarded as a process by which a saturated soil is compressed under a long-term load accompanied by a reduction in porosity with expulsion of water. In contrast, soil compaction is considered to be a process in which an unsaturated soil is compressed by a load applied for a short time with no expulsion of water. However, such a distinction in compacted soil management does not seem practical.

Stunted plant growth, shallow and malformed plant roots, standing water, formation of large clods after tillage, and physically dense soil are the signs of soil compaction. Soil compaction has been described as one of the five threats to sustained soil quality by the EU Soil Framework Directive (Commission of the European Communities 2006). There is about 68.3 M ha compacted soil globally, which accounts for 4 % of anthropogenic soil degradation (Oldeman et al. 1991). In Europe, compaction accounts for about 17 % of the total degraded area. It is a complex problem in which machine/soil/crop/weather interactions play an important role and may have economic and environmental consequences for world agriculture (Soane and van Ouwerkerk 1995). Soil compaction is a global problem commonly associated with mechanized agriculture. It has caused yield reductions of 25–50 % in some regions of Europe (Eriksson et al. 1974) and North America and between 40 and 90 % in West African countries (Charreau 1972; Kayombo and Lal 1994). On-farm losses through soil compaction in the USA have been estimated at US$1.2 billion per year (Gill 1971). Oldeman et al. (1991) suggested that compaction is by far the most important type of physical deterioration of agricultural soils.

Fig. 2.1 Soil moisture level and susceptibility to compaction



There are two types of soil compaction: shallow and deep. Shallow compaction occurs near the soil surface and can be broken up by normal tillage. Surface crust and hardsetting are a kind of shallow soil compaction. Deep compaction is caused mainly by axle load. Compacted layers below the plow depth are known as tillage pans or plow pans. Van den Akker and Schjonning (2004) suggested that extensive areas have undergone subsoil compaction due to use of heavy farm machineries. Deep compaction may extend as far as 75 cm below the surface. It is extremely difficult to correct since it is below the normal tillage zone. Deep compaction reduces water and air storage in the deeper part of the soil profile and hamper the growth of all types of crops. Developing full root system of deep-rooted crops such as corn and alfalfa is hampered by deep compaction.

2.2.3.1 Factors Affecting Soil Compaction

Any process that reduces pore space causes soil compaction. Farm machineries including tractors and other implements can compact soil when traveling across a field. Livestock and even human can compact soil. The extent of soil compaction depends on many factors: soil physical conditions as well as weight and design of the load on soil, distribution of the load over the number of axles and tires, and, to a lesser degree, number of trips made over the field. Soil texture, structure, organic matter, and water are important soil factors that determine susceptibility of soil to compaction. Soils made up of particles of about the same size compact less than soil with a variety of particle sizes. Smaller particles can fill the pores between larger particles resulting in a denser soil. A sandy loam soil is more easily compacted than a sandy or clayey soil. A soil with a higher level of organic matter generally has a better structure. Hard, dense, low organic matter soils suffer more from compaction than loose, friable, high organic matter soils. A dry soil is not easily compacted. Soil compaction increases with the increase in soil water content during tilling. A soil saturated with water resists compaction because water occupies the pores and water is not compressible. Soils are easily compacted at soil water contents slightly higher than field capacity. Figure 2.1 shows the relationships of soil water with soil compaction.

Fig. 2.2 Heavy farm machineries that can seriously compact soil

The major cause of soil compaction is the traffic over the fields and that too with heavy farm machineries. Farm tractors, combines, and other equipment are large and heavy. Tractors weigh up to 20 t and liquid manure spreaders can weigh as much as 15 t when fully loaded.

Heavy Farm Machineries and Soil Compaction

Farm operations in industrial countries are completely mechanized. Tillage, seeding, planting, harvesting, and processing are done with heavy machineries. Agricultural tractors and other implements such as combines are used for tilling, planting, and harvesting. These 20–200 HP gasoline and diesel 2-wheel drive and 4×4 machines usually have power steering, dual-range transmissions, and two-speed live power take off (Fig. 2.2). Other farm implements include disks, plows, tillers, scrapers, and diggers. Tractors alone have weights of about 20 t. These heavy agricultural machineries result in much irreversible damage to the soil. Soil compaction, induced by wheel traffic from large agricultural equipment, has been shown to extend well below the depth of plowing, reducing crop growth and yield in many situations. Estimates suggest that the area of soil degradation due to compaction in Europe may exceed 33 M ha.

Higher tire inflation pressures are used to support such heavyweights. This heavyweight gives higher pressures on the soil surface. Soil pressure is approximately equal to inflation pressure with radial tires, while with bias-ply tires there is increased soil pressure at the tire’s edge because the sidewall carries part of the load. Both radial and bias-ply tires are used on tractors, combines, and implements. Dual tractor tires, compared to single tires, decrease pressure on soil, but increase the affected area. As little as 4 psi pressure can cause soil compaction, duals or even high-flotation tires do not prevent compaction; they merely change the distribution. A pickup truck will impose about 50 psi pressure on the soil, a liquid manure spreader 70–90 psi, and a 500 kg cow approximately 40 psi. Number of passes of equipments over the field is important regarding soil compaction. Even one pass over a field under poor conditions can cause significant damage. The first pass of a wheel causes 80 % of the potential compaction. Subsequent passes cause additional, but progressively less, compaction. After four passes, the additional compaction becomes very small.

Tillage can either create or help to alleviate soil compaction. Tillage operations break up soil into smaller particles. Excessive tillage destroys the structure that provides desirable pore space. Some tillage equipment, such as moldboard plows, may aerate the soil and increase percolation at the surface while creating a compacted layer just below tillage depth. Such a layer is called a “plowpan” or “hard pan.” Disks can also produce a hard pan just below tillage depth while overtilling the soil near the surface, especially where multiple passes are made.

2.2.3.2 Effects of Soil Compaction

The effects of soil compaction on soil properties and processes have been reviewed by Soane et al. (1982), Lipiec and Stepniewski (1995), and Alakukku (1999). Soil compaction has been found to affect almost all physical, chemical, and biological properties and processes of soil to variable extents. Soil compaction modifies the pore size distribution of mineral soils, mainly by reducing the porosity and especially the macroporosity (diameter >30 μm; Eriksson 1982; Ehlers 1982). Besides the volume and number of macropores, compaction also modifies the pore geometry, continuity, and morphology. Soil compaction has negative impacts on many soil properties related to soil working, drainage, crop growth, and the environment. Compaction due to field traffic increases the bulk density (Arvidsson 1998), shear strength and penetrometer resistance (Blackwell et al. 1986) of soils, limiting root growth and increasing the draft requirement in tillage. Soil compaction reduces infiltration (Pietola et al. 2005) and saturated hydraulic conductivity (Alakukku et al. 2003). Soil compaction reduces CO_2 and O_2 exchange (Simojoki et al. 1991). Drainage problems appear due to loss of permeability by compaction in the subsoil. Soil compaction may lead to waterlogging. Poorly drained soil may also dry slowly, reducing the number of days available for field operations. The reduction in drainage rate attributed to soil compaction can be expected to increase the emissions of greenhouse gases from soil (Ball et al. 1999), for instance, by increasing denitrification. Compaction increases surface runoff and topsoil erosion (Fullen 1985). By affecting soil properties and processes, soil compaction influences crop growth, yield, and the use efficiency of water and fertilizers. Soil compaction reduces yield (Hanssen 1996), crop water use efficiency (Radford et al. 2001), and nutrient uptake (Alakukku 2000).

Typical responses of plants to soil compaction include low seed germination, reduced seedling emergence, reduction of number and length of roots, restriction of downward penetration of the main root axes, decrease in leaf thickness, increase in shoot-to-root ratio, and decrease in crop grain yield (Fageria et al. 2006). The degree of restriction of root growth in compact soil depends also on the species and the age of the plants (Masle 2002). Inhibited plant growth is mostly attributed to reduced rooting volume (Grzesiak et al. 2002; Masle 2002). The restrictive effect of soil compaction can be physical and physiological constraints to overall plant growth and yield through poor development of root system (Iijima et al. 1991; Grzesiak et al. 2002) since high soil impedance influences root elongation and proliferation. Acquisition of water and mineral nutrients is primarily determined by dimension of root zone and distribution of roots.

2.2.3.3 Management of Compacted Soils

The problems associated with soil compaction are low infiltration, high runoff, poor root penetration, low available soil moisture, and poor aeration. So, the principles of compacted soil management should include:

- Avoiding to make the soil more compact
- Avoiding excessive tillage
- Avoiding to till the soil when it is too moist (at or slightly higher than field capacity)
- Making and keeping the soil loose and friable
- Increasing infiltration
- Reducing runoff
- Reducing evaporation

The techniques may involve reduced tillage, building up of organic matter, controlled trafficking, and mulching.

Subsoiling: Subsoiling with heavy duty shanks to depths of 35–60 cm or even deeper and usually spaced about 30–75 cm apart has been tried with mixed results. Subsoiling is an expensive operation requiring about 30 hp per shank. Its greatest benefits appear to occur in the southeastern USA, where soil is subject to little or no freezing. Short-term benefits have been measured, but some scientists think that subsoiling may, in the long run, make compaction worse. Subsoiled fields may recompact to a greater degree from normal operations than without any such subsoiling treatment. Apparently, subsoiling reduces the soil's capacity to support loads of equipment without compacting.

Reducing Tillage: The less the soil is tilled, the less it is compacted. Tillage contributes to the breakdown of soil structure by compressing and breaking the aggregates, necessary for aeration and root growth. Tillage also increases decomposition of organic matter needed for aggregate stability. Organic matter depletion goes hand-in-hand with structure depletion. Reduced tillage leaves higher amount of plant residues on the surface which prevent surface sealing.

Controlled Traffic Farming (*CTF*): Controlled trafficking is adopted in Australia to reduce the risk of soil compaction since the early 1990s involving production of grain crops under a production system where the high-powered tractors, large harvesters, and heavy trucks all have the same track settings of 3 m. In the field, the width of the combine harvesters and planters is matched at perhaps 9 m, allowing the 3 m tracks to be centrally spaced at the same distance. The advantages are the tractor can be less powerful because the wheels run on packed soil while only minimum or no tilling is required on the non-compacted beds in between; nor is there any need to subsoil every few years. In CFT, water penetration is better, there are no ruts or ridges left over from previous operations, the fuel saved is substantial, and the crops yield much better because of zero compaction.

Building Up of Organic Matter: Organic matter improves soil structure, decreases soil bulk density, and increases porosity. Organic matter resists compression and consolidation of soil. For building up of soil organic matter and keeping the soil loose, generous additions of farmyard manures, poultry manures, composts and municipal biosolids, and adopting green manuring and crop rotations are needed.

Mulching: Mulching increases infiltration and decreases runoff and evaporation. Mulching conserves soil moisture. Several materials are used as mulch. Types of organic mulches are compost, composted manure, grass clippings, newspaper, straw, shredded leaves, straw, etc. Types of inorganic mulches are gravels, pebbles and crushed stones, plastic sheets, woven ground cloth, aluminum-coated plastic foil, and ground rubber tires. Organic mulches are natural and cheap materials, and along with reducing compaction, they improve soil fertility. Mulches protect soils, increase infiltration, improve water-holding capacity, reduce weed growth, and prevent soil compaction. Organic mulches are gradually decomposed and release plant nutrients. Mulched plants have more roots than plants that are not mulched.

2.3 Waterlogging

Waterlogging is a state of saturation of soil with water for a prolonged period. Soils may, sometimes, be saturated with water for a very short time, say some hours, as during flash floods in piedmont areas. This temporary stagnation is not regarded as waterlogging. This water drains away easily. Waterlogging is a condition of soil when draining excess water is difficult and requires time, labor, planning, and energy. Waterlogging develops due to many different causes, including natural conditions and human activities. Natural conditions include heavy rains, low lands, clay soils, flooding, and presence of impervious subsoil. Human activities include faulty irrigation, inadequate drainage, surface sealing, and deep soil compaction by tillage implements. Wetlands are permanently waterlogged. Waterlogging also occurs when the groundwater table rises to the root zone and remains there for a considerable period of the year (Michael and Ojha 2006). Waterlogging may also occur as a form of standing water in the farm, which does not lower with time (Murty 1985). Worldwide, about 10 % of all irrigated land suffers from waterlogging. As a result, productivity has fallen about 20 % in these croplands.

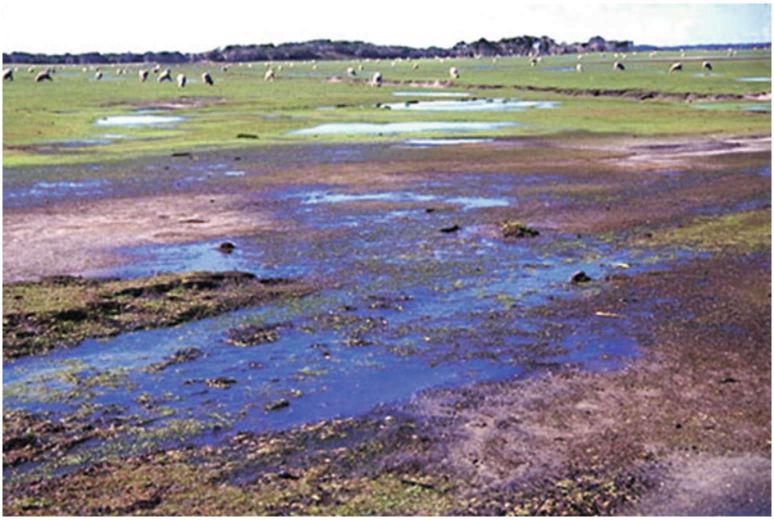


Fig. 2.3 Waterlogging in agricultural land (Photo courtesy of Dr. Richard Doyle)

2.3.1 Causes of Waterlogging

Natural conditions responsible for waterlogging include heavy rains, low lands, clay soils, flooding, and presence of impervious subsoil. Human activities that create or aggravate waterlogging include faulty irrigation, inadequate drainage, and surface sealing as well as deep soil compaction by tillage implements.

Wetlands are naturally and permanently waterlogged, but we are primarily concerned with arable lands because waterlogging hampers the growth of most crop plants. The most important causes of waterlogging are the poor drainage due to compaction (Fig. 2.3) and the rise of the groundwater table. Compaction including surface sealing, crusting, hardsetting, and deep compaction results from organic matter depletion, structure deterioration, dispersion, compression, and consolidation. These processes occur due to inappropriate tillage and applied pressure by the heavy load of farm machineries (Sect. 2.2.3.1). Soil compaction reduces hydraulic conductivity, thereby reducing both infiltration and percolation. Rainwater or irrigation water cannot move to the groundwater table deep in the regolith and drain away. In some situations, water accumulates and the groundwater table rises toward the surface of the soil in absence of significant base flow. Usually, a small area is irrigated in small farm holdings, so that over-irrigation is done and high percolation compels the groundwater table to rise. Rising groundwater table prevents root respiration and restricts their functioning. Elevated groundwater table may also create salinity and reduce crop yield.

In soils where there is a deep compaction, plant roots are confined to the loose surface soil, and excess irrigation water cannot pass readily through the impervious layer. It creates shallow root system and leads to waterlogging. In some cases, an impervious stratum may occur below the top layers of pervious soils. In this case also, water seeping through the pervious soils will not be able to go deep and, hence, quickly results in waterlogging. In the absence of satisfactory natural drainage, well-planned artificial drainage systems are needed to be installed. Often, drainage systems do not work satisfactorily, neither they are sufficiently deep, nor they are adequately frequent.

2.3.2 Effect of Waterlogging

Waterlogging reduces aeration. After some days of flooding, the oxygen content in soil may completely deplete. Growth of crop plants and yield are reduced by altered physiological processes due to low oxygen and reduced root respiration in waterlogged soils. Often, waterlogging is responsible for crop failures. In addition, waterlogging does not allow following a definite crop calendar. The normal cultivation operations, such as tilling and plowing, cannot be easily carried out in wet soils. Plowing, sowing, and planting are delayed by waterlogging. Waterlogging converts many croplands into wastelands. Rising water table brings dissolved salts in groundwater to the surface soil. Water continuously evaporates leaving the salts in soil. Thus, a high water table creates soil salinity, which is difficult to reclaim. Soil salinity reduces productivity. Many soils have become saline and out of cropping due to faulty irrigation in different countries, including Pakistan.

2.3.3 Control of Waterlogging

Waterlogging can be controlled only if the passage and quantity of water into the subsoil is reduced; the rising of the groundwater table is prevented, and the drainage of excess water is improved. The inflow of water into the underground reservoir may be reduced by reducing the intensity of irrigation. The outflow of water may be increased by deep and frequent interceptor and field drains. The groundwater table must be maintained well below the root zone. Generally, keeping the water table at least about 3 m below the ground surface is desirable. This can be achieved by different drainage systems. Seepage water can be prevented by interceptor drains. Stagnant water can be removed by field drains, surface or subsurface. The drains must be regularly monitored. Many soils become waterlogged by flooding from canals and rivers. Small- and large-scale embankment has been successfully employed in different regions.

2.4 Lowering of Groundwater Table

Water that is located beneath the earth's surface in pores and fractures in soil and rock is called the groundwater. The highest level of this underground water is known as the groundwater table. The water table is actually the boundary between the unsaturated and saturated zones. It represents the upper surface of the groundwater. The water level found in unused wells is often the same level as the water table. Soil pore spaces in soil become completely saturated with water up to the groundwater table. The entry of water to the saturated zone beneath the ground is called the groundwater recharge. Groundwater recharge is a hydrologic process where water moves downward from surface water to groundwater. This process usually occurs in the vadose zone below plant roots and is often expressed as a flux to the water table surface. Recharge occurs both naturally and through anthropogenic processes. The sources of recharge to a groundwater system include both natural and human-induced phenomena. Natural sources include recharge from precipitation, wetlands, lakes, ponds, and rivers (including perennial, seasonal, and ephemeral flows), and from other aquifers. Human-induced sources of recharge include irrigation losses from canals and fields, leaking water mains, sewers, septic tanks, and over-irrigation of parks, gardens, and other public amenities. Recharge from these sources has been classified as direct recharge from percolation of precipitation and indirect recharge from runoff ponding. When groundwater recharge exceeds discharge, the groundwater table rises upward. When groundwater pumping and discharge exceeds recharge, there is the lowering of the groundwater table.

Removal of water from the groundwater is known as the groundwater discharge. Some, and often a great deal, of the water flowing in rivers comes from seepage of groundwater into the streambed. Groundwater contributes to streams in most physiographic and climatic settings. The proportion of stream water that comes from groundwater inflow varies according to a region’s geography, geology, and climate. Movement of groundwater in these directions is known as groundwater discharge. It is the removal of water from groundwater. A plenty of groundwater is also withdrawn from shallow and deep tube wells in urban and agricultural areas. The most severe consequence of excessive groundwater pumping is that the water table is lowered. This poses a threat to the future shortage of water for urban and agricultural use. A related effect of groundwater pumping is the lowering of groundwater levels below the depth that streamside or wetland vegetation needs to survive. The overall effect is a loss of riparian vegetation and wildlife habitat. The basic cause of land subsidence is a loss of support below ground. When water is taken out of the soil, the soil collapses, compacts, and drops. This depends on a number of factors, such as the type of soil and rock below the surface. Thus, the water table should be maintained at a desirable depth beneath the soil surface. A shallow water table restricts root growth and elongation by limiting oxygen. Large-scale lowering of water table has been considered as an important cause of desertification.

2.5 Subsidence of Organic Soils

Subsidence is the fall of the land level due to compression, consolidation, compaction, and oxidation of organic matter, particularly as a result of draining and cultivation of peat soils (Histosols). Subsidence has long constituted a serious problem in the reclamation and development of peat lands in many parts of the world. Draining brings the surface soil layers above the submergence level, which enhances aeration. So, reclamation of peat soils will always accompany oxidation of these layers. Experiences in the Netherlands show that reclamation of peat areas in the Middle Ages have gradually subsided from 0.5 m above mean sea level to 1–2 m below. Land subsided about 2 m over a period of 8–10 centuries in spite of a continuously shallow drainage (Nieuwenhuis and Schokking 1997). Some 85 % of this subsidence can be ascribed to the oxidation of organic matter, which will continue at a rate of 5 cm year^{-1} (Schothorst 1982). Land subsidence is the most commonly observed response of Histosols to drainage for agricultural purposes. Worldwide subsidence rates in drained peaty areas vary from less than 1 cm year^{-1} to more than 10 cm year^{-1}. The arable organic soils experienced an average subsidence rate of 2.5 cm year^{-1} between 1924 and 1978 (Ingebritsen et al. 1999) in Florida. Histosols in California settled at a rate of up to 8 cm year^{-1} between 1922 and 1950 (Rojstaczer and Deverel 1995). Records of land subsidence in Malaysia reveal that the subsidence rates decreased from 12 cm year^{-1} over the period 1960–1974 to 6.4 cm year^{-1} in the following 14 years and to 2 cm year^{-1} thereafter (Wosten et al. 1997). Stephens et al. (1984) recognized at least five sources of organic soil subsidence following drainage: shrinkage due to desiccation, consolidation, wind and water erosion, burning, and biochemical oxidation. The latter has been found to be the dominant cause of land subsidence in temperate and tropical peat soils (Andriesse 1988; Deverel and Rojstaczer 1996). In areas with low-bearing capacities, buildings and structures have to be built on pile foundations. Subsidence will change the relative elevation of piled buildings and structures with respect to the surrounding area. The areas surrounding these buildings and structures will have to be raised from time to time by the addition of earth or other fill material. Special measures have to be taken in connecting utilities (power lines, water mains, etc.). On soils with soft clay or peat layers, the design height of embankments has to be corrected to take the future subsidence into account; otherwise, the safety requirements may not be met. Segeren and Smits (1980) suggested that clay content, depth of peat layer, loss of water due to drainage, water table, and load of machineries affect the rate and extent of subsidence. Clayey sediments lose more water than sandy sediments. As a consequence, clay soils will subside more. The subsidence caused by the shrinkage of different soil layers at a given clay content decreases with depth. The greater the thickness of the compressible layers, the greater will be the subsidence. Cultivation with heavy machineries compacts the peat soils considerably.

2.6 Desertification

Desertification is a process of land degradation which occurs mainly in the arid and semiarid regions due to human actions on ecosystems combined with adverse climatic conditions. Dregne (1977) defined, “Desertification is the impoverishment of terrestrial ecosystems under the impact of man. It is a process of deterioration in these ecosystems that can be measured by reduced productivity of desirable plants, undesirable alterations in the biomass and the diversity of the micro and macro flora and fauna, accelerated soil deterioration, and increased hazards for human occupancy." The formal definition of desertification adopted by the United Nations Convention on Desertification (UNCCD) in 1994 is, "Land degradation in arid, semi-arid, and dry sub-humid areas resulting from various factors, including climatic variations and human activities." Due to desertification, forests, grasslands, shrublands, and croplands are converted to desert-like lands. Desertification has gained growing attention of the international community since the 1970s about its devastating and destabilizing potential on the natural environment and the human society. Based on United Nations estimates, the phenomenon has interested nearby 70 % of arid lands, amounting to about 30 % of the world cultivable land. The problem is particularly severe in Africa and in several developing countries in Asia, South America, and the Caribbean, but it interests also the USA, Australia, and Southern Europe (especially Greece, Spain, Portugal, and also Italy) (Eswaran and Reich 1998; Perini et al. 2009). According to OECD, the main processes of soil degradation are due to erosion, submersion, acidification, salinization, soil compaction, surface crusts and compact layers along the profile, loss of organic matter, deterioration of the soil structure, accumulation of toxic substances, as well as loss of nutrients (Perini et al. 2009). Desertification affects over 2 billion people and around 100 countries across all 5 continents (Bied-Charreton 2008). Ten million ha productive land with 24 billion tons of topsoil is lost every year due to desertification. The principal worldwide problems caused by desertification are loss of biological productivity, loss of economic productivity, and loss of complexity in the landscape (UNCCD 1994).

2.6.1 Natural Versus Human-Induced Desertification

Causes of desertification are complex, frequently local, and vary from one part of the world to another. The severity of impact also varies with less-developed countries experiencing greater human misery. The causes and processes of desertification are much debated among the experts of many different disciplines such as geography, ecology, environmental science, economics and sociology. According to some scientists, desertification is a natural phenomenon occurring due to irregular fluctuation of short- and long-term drought, as experienced in African Sahel (Hill and Peter 1996). Other scientists believe that desertification is a human-induced phenomenon resulting from pressures on land resources, unwise development policies, and misuse of land (Graetz 1996). Regions vulnerable to desertification are undeveloped, poverty stricken, and poor in biomass resources. Soil and vegetation in the arid and semiarid regions are naturally fragile. Still, human pressure on vegetation and land is enormous, but the support of technology, soil, and climate is inadequate. Culet (2002) suggested that overgrazing, deforestation, and other agricultural activities may respectively contribute 35, 30, and 28 % to desertification.

Unsustainable agricultural practices including excessive use of chemicals (fertilizers, pesticides, and herbicides), inappropriate technologies, wrong choice of crops, inadequate and inefficient irrigation practices associated with salinity, and over abstraction of groundwater have led to problems of desertification in vulnerable regions. There are about 7.1 million km^2 of land at low risk of human-induced desertification, 8.6 million km^2 at moderate risk, 15.6 million km^2 at high risk and 11.9 km^2 at very high risk (Eswaran et al. 2001).

2.6.2 Convention to Combat Desertification

Following a severe drought in sub-Saharan Africa, a United Nations Conference on Desertification was convened in 1977 in Nairobi, Kenya. This was probably the first global conference on land degradation. Although it was called a conference on desertification, land degradation and its control was the sole subject. In 1990, at a meeting in Nairobi called by the United Nations Environment Program, desertification was formally defined as land degradation in the dry lands. At the 1992 United Nations Conference on the Environment and Development, participating governments approved a resolution to establish an international agreement to combat desertification. The title of the formal agreement is the United Nations Convention to Combat Desertification in Those Countries Experiencing Serious Drought and/or Desertification, Particularly Africa. The short title of the ratified agreement is the Convention to Combat Desertification (CCD). However, much has not been achieved regarding the reversing of desertification situation.

2.6.3 Mechanism of Human-Induced Desertification

Desertification caused by human influences has a long historical record; there is evidence of such damage caused around the Tigris and Euphrates rivers in ancient Mesopotamia. Human influences on desertification include loss of vegetation by deforestation, shifting cultivation, overgrazing, depletion of groundwater, frequent burning, and compaction of the soil. These factors affect desertification which is a process of continuous and gradual ecosystem degradation, during which plants and animals and other natural resources such as water and soil are stressed beyond their ability to adjust to changing conditions. The physical characteristics of land undergoing desertification include progressive loss of mature, stabilizing vegetation from the ecosystem, or loss of agricultural crop cover and a resulting loss of unconsolidated topsoil.

Early human pastoralists living in semiarid regions moved their small groups of domestic animals in response to food and water availability. Such regular stock movement prevented overgrazing of the fragile plant cover. Settled human habitation increased the risk of overgrazing by livestock around the settlements.

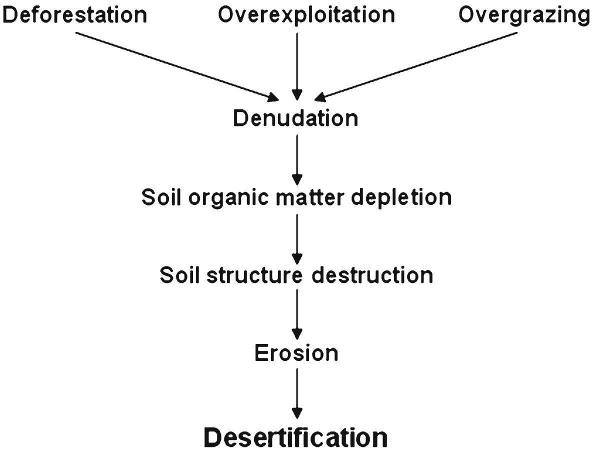


Fig. 2.4 Desertification scheme

Overgrazing is the major cause of desertification worldwide. Cultivation of marginal lands, lands on which there is a high risk of crop failure and a very low economic return, increases the risk of desertification in many areas. Destruction of vegetation in arid regions for fuelwood and burning of vegetation are responsible for denudation in many places. Incorrect irrigation practices in arid areas can cause salinization, which can prevent plant growth. Organic matter of soil is depleted due to reduced biomass production and enhanced decomposition. Soil structure is destroyed and soil particles are removed by water and wind erosion. Continuous erosion exposes the subsoil which lacks the capacity of supporting sufficient vegetation. Wind and water erosion leads to development of gullies and sand dunes across the deflated land surface. A hypothetical desertification scheme is presented in Fig. 2.4.

Study Questions

1. What do you mean by physical deterioration of soil? What are the types and processes of physical deterioration of soil? Distinguish among surface sealing, crusting, hardsetting, compression, consolidation, and compaction. Soil compaction is a problem associated with mechanized agriculture – Discuss.
2. What are the differences among flooding, submergence, water saturation, and waterlogging? How does human activity aggravate waterlogging? What are the consequences of waterlogging of soil?
3. What is groundwater? How does interaction take place between groundwater and surface water? What do you mean by recharge and discharge of groundwater? What are the sources of groundwater recharge? What are the consequences of lowering of groundwater table?
4. What are the characteristics of peat soil? Discuss subsidence of peat soils due to their draining and reclamation.
5. What is desertification? Distinguish between denudation and desertification. Explain the processes of desertification.

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Chapter 3
Soil Erosion by Water

Erosion is a natural process of detachment of soil particles and their transport and deposition at distant places by natural agents such as water, wind, glacier, and gravity. Detachment of soil particles from aggregates primarily by raindrops and flowing water and their transport by runoff water are involved in soil erosion by water. Natural erosion is considered as normal erosion and is usually of little concern from soil quality point of view because its rate is low and soil loss can be naturally compensated by soil formation. Human actions such as deforestation, overgrazing, over tilling, and shifting cultivation have accelerated soil erosion beyond the tolerance limit. A tolerance range of 2–11 t ha^{-1} $year^{-1}$ depending on soil types is accepted in the USA. There are places and situations where erosion rates are much higher than this limit, even as high as 100 t ha^{-1} $year^{-1}$. The principal types of soil erosion by water are splash erosion, sheet erosion, interrill erosion, rill erosion, gully erosion, landslides, and stream erosion. Soil erosion has on-site and off-site effects. The on-site effects include loss of soil, loss of organic matter and nutrients, damage to growing crops, exposure of plant roots, and decline in soil fertility and productivity. The off-site effects are burrowing of crops and installations, siltation of reservoirs, eutrophication of ponds and lakes, pollution of water, etc. Several agronomic and engineering practices are employed for the control of water erosion. These are no-tillage, minimum tillage, mulching, strip cropping, contour cropping, contour strip cropping, and terracing, but several methods are needed to be integrated for an efficient soil erosion control.

3.1 Geological and Accelerated Erosion

Erosion is the detachment of materials from earth surfaces such as rock and soil from its original assemblage and position and transport to other places by various agents, including water, wind, glacier, and gravity. Erosion has both on-site and off-site effects. Earth materials from the mountains of the Himalayas are being torn out

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and carried by river systems to the Gangetic Delta through geological time. The process brings about changes in landforms of both the places. The Himalayas are weathering away and the deltas are accumulating alluviums, and the land level rises above the sea level. In many cases, frost and high temperature separate pieces of weathered rock, and the loose material moves downhill to form piles of hillside waste, debris cones, outwash fans, and other formations (Zachar 1982). This is geological erosion occurring in almost all natural surfaces through natural forces. However, erosion may be of different types, such as rock erosion, land erosion, and soil erosion, depending on the surface concerned. Erosion may be divided into natural or geologic and human-induced or accelerated erosion.

Soil erosion involves two processes: detachment of particles from soil aggregates and transport of the particles by water or wind. Soil erosion in undisturbed landscapes by natural forces is called normal erosion, natural erosion, or geological erosion which occurs almost silently and often leaves no signs. Natural erosion is not of much concern because the amount of soil loss is readily compensated by the natural processes of soil formation. But human actions for the exploitation of land, water, vegetation, and soil resources tremendously accelerated the extent and intensity of soil erosion. This is known as accelerated soil erosion. Generally accelerated erosion is considered to be soil erosion proper. Accelerated erosion occurs usually at an alarming rate that reduces soil quality and crop yield on-site and damages land, water, and installations off-site.

3.2 Causes of Water Erosion

Causes of soil erosion include deforestation, overgrazing, biomass burning, shifting cultivation, mechanized logging, cultivation up- and downslope, continuous mono cropping, and compaction of soil by stock and machinery. Deforestation, overgrazing, shifting cultivation, and mono cropping have been discussed in connection with causes of soil degradation in Chap. 1. Compaction was discussed in detail in Chap. 2. The reader may extract a good account of the causes of soil erosion from these chapters. It is worth mentioning here that water erosion is a prominent problem of the sloping lands. Disturbing the slopes for seedbed preparation and cropping is very risky. General soil conservation practices such as minimum tillage and mulching help reduce the risks. But slopes steeper than 20° should be left under natural conditions. Satisfactory forests and grasses can develop there if left undisturbed. Such slopes are very susceptible to water erosion. Cultivating up and down that is along the slope produces severe water erosion.

3.3 Types of Water Erosion

Water erosion is caused by water – water that comes in rain and runs off the land as overland flow or streamflow. At the initial stage, soil particles are detached from aggregates by the impact of falling raindrops or flowing water, which is followed by

Fig. 3.1 Splash erosion (Photo courtesy of Dr. Lasikaamarasena)

transport of the detached particles by runoff water. Runoff water laden with suspended particles also detaches more soil particles in its way across the surface. Soil erosion is a process of soil loss, particularly from the surface, but sometimes a large mass of soil may be lost, as in landslides and riverbank erosion. Kohnke and Bertrand (1959) classified water erosion into sheet erosion, internal erosion, and channel erosion. Channel erosion was further divided into rill erosion, gully erosion, and stream erosion. However, the following four types of water erosion are generally recognized: splash erosion, sheet erosion, rill erosion, and gully erosion. Splash and sheet erosion are sometimes called interrill erosion.

3.3.1 Splash Erosion

At the start of a rain event, falling raindrops beat the soil aggregates, break them, and detach soil particles. These particles clog the large soil pores and, thus, reduce the infiltration capacity of the soil. Water cannot enter the soil, and soon a thin film of water covers the ground. Further, raindrops beat the water and splash the suspended soil particles away. Soil particles are transported to some distance by the splashing. The splashed particles can rise as high 60 cm above the ground and move up to 1.5 m from the point of impact. Processes of splash erosion involve raindrop impact, splash of soil particles, and formation of craters (Ghadiri 2004). Actually, splash erosion (Fig. 3.1) is the beginning of other types of soil erosion, particularly sheet erosion.

Fig. 3.2 Severe sheet erosion has exposed tree roots in a teak forest (Photo courtesy of Dr. Animesh Biswas)

3.3.2 Sheet Erosion

When a thin layer of soil is removed by raindrop impact and shallow surface flow from the whole slope, it is called sheet erosion. It removes the finest fertile topsoil with plenty of nutrients and organic matter. It is the most dangerous type of soil erosion because it occurs gradually and almost silently leaving little or no signs of soil removal.

Sheet erosion involves the removal of a more or less uniform layer of soil over the whole slope of the land (Fig. 3.2). Soil particles are detached primarily by raindrops and secondarily by frost, hooves of farm animals, tillage, and mechanical action of farm machines. Detached particles are transported by runoff water as overland flow. Sheet erosion is more uniform and gradual, as the surface becomes smoother. However, water may still accumulate even on the smoothest slope. The intensity of accumulation of runoff water depends on the height of the water stream, the coarseness of the surface, vegetation, or crop distribution. Sheet erosion removes deeper layer of soil gradually, if allowed to proceed unhindered and the subsoil is exposed over a large area. The subsoil is usually of different texture and color and is more compacted. However, slopes are often not so uniform over the whole area, and water accumulates in tiny channels, so that the surface is crisscrossed by discontinuous rillets. It is then known as the interrill erosion (Fig. 3.3).

Fig. 3.3 A sloping soil surface affected by interrill erosion (Photo courtesy of USDA-NRCS)

In addition to soil particles, sheet erosion removes (i) organic residues accumulated on the surface soil and (ii) soluble and easily dissoluble matter, matter made soluble by weak acids in rainwater. Zachar (1982) states that sheet erosion represents microerosion, that is, the eroding and washing of the soil to produce small-scale forms which may encompass raindrop erosion, laminar erosion, rillet erosion, and layer erosion. The first phase of sheet erosion, specific with regard to form, is soil removal by raindrop action – raindrop erosion. In raindrop erosion, the surface is acted upon selectively so that small holes, micropyramids, and other forms occur, raindrop erosion thus becoming a part of pedestal erosion, pinnacle erosion, etc. The second subtype in sheet erosion is laminar erosion. It occurs in any flow of water on an inclined soil surface where the kinetic energy of the water is small and only the finest soil particles are consequently washed away in a strongly selective manner. By virtue of the accumulation of sheet runoff water, rill erosion develops, causing small rills with the dimensions of a few centimeters diameter in cross section, and with a depth not exceeding that of the arable layer. The rillets that develop in rows and furrows, with the effect of increasing their dimensions and conspicuousness, are removed during cultivation. In this form of erosion, soil and particles displaced by water may be intensively separated and sorted. In layer erosion, the soil is washed away neither in laminae nor in rillets, but in a layer up to several meters wide and 10–25 cm deep from a tilled surface, that is, in apparent strips from which the topsoil has been entirely removed.

Fig. 3.4 Rill erosion (Photo courtesy of USDA-NRCS)

3.3.3 Rill Erosion

When rainfall exceeds the rate of infiltration, water accumulates on the surface, and if the land is sloping, it moves along the slope. On gently sloping lands, with standing crops or in fields that have been recently tilled, moving water concentrates along tiny channels called rills (Fig. 3.4). Rills are less than 30 cm deep. The cutting action of flowing water detaches soil particles, and runoff water carries them away. The amount of soil loss may be high, but the small channels do not usually interfere with tillage implements. The rills may be leveled by normal tillage operations. Rill erosion is often the initial stage of gully erosion.

Rill erosion is largely caused as a result of large amounts of material that are released and transported for variable distances in concentrated areas. On the other hand, the flow of water over the surface has a smaller effect on soil detachment, but a larger transportation effect. Yet flowing water, especially on tilled land, can become the agent of transport of particles loosened mechanically, chemically, or by means other than the water flow itself, and therefore it is a phenomenon of great importance from the point of view of total soil losses (Zachar 1982). Water concentrates in places over the field due to reduction in infiltration, increase in precipitation, and surface roughness of the land. Water concentrates along tillage lines, rows of crops, impedance by exposed roots, around clods, etc. and from shallow and narrow channels known as rills. As this gathering of water proceeds, the total amount of water remaining the same, the depth of the water increases, together with the velocity, kinetic energy, and detaching as well as carrying capacity of the water. At high precipitation intensities, there is greater clogging of pores, and the proportion of precipitation water making up the surface flow and the numbers of particles separated from the soil by raindrops both increase.

Greater predominance of rill erosion may be found on steeper slopes with impermeable soil material consisting of younger sediments which are susceptible to erosion. As is seen usually, rill erosion prevails and affects the whole length of the slope, which means that precipitation water, as soon as it reaches the soil, flows away through the dense network of rills, virtually cutting the slope into thin plates. Occurrence of splash erosion or sheet erosion is not seen, but there was some erosion of these forms too. Similar phenomena occur on steep slopes, even on impermeable loamy clay material. On impermeable or still heavier and more resistant material, rill erosion forms ridges which are separated by sharply cut rillets and gullies. The rillets are occasionally so narrow that they resemble cracks, and it could be termed crack erosion. On steep slopes composed of material of varying resistance, vertical openings are formed, and these soon develop into tunnel erosion or hollow erosion, separating the washed forms into isolated pipes, etc. Where the material is more homogeneous and the incline less steep, rilling prevails. If the material is more coarse grained and less resistant, flowing water carries the soil along rapidly and creates triangular or trough-shaped forms with respect to the cross section of the channel. In such cases, the lengths of the rills are greater, but the interrill lamellae are thinner, and the edges sharper. The more coarsely grained and more permeable the material, the less pronounced is the channeling, until finally the rills are widely shaped, and resemble more the form of moderately undulating depressions, even on very steep parts of the eroded slope. On permeable, coarse-grained, non-resistant fluvioglacial deposits, on the other hand, shallow, rapidly growing rills develop with an immense production of silt. Here, the action of flowing water is the predominant force. These forms are transitional toward gully erosion. Rill erosion usually begins to appear in the lower part of the slope. This is true especially when the source of the water is thawing snow or precipitation of low intensity. As soon as the intensity of the rainfall increases, the intensity and velocity of surface runoff both increase also, and consequently the proportion of the total erosion due to rills becomes greater, depending on the permeability of the soil.

3.3.4 Gully Erosion

Gullies are large channels deeper than 30 cm. Gullies develop when large quantities of water accumulate and run through a single channel with high speed in relatively steep slopes (Fig. 3.5). Gullies may also develop by the gradual deepening of rills. There are two types of gullies: ephemeral and permanent. Ephemeral gullies form shallow channels that can be readily corrected by routine tillage operations. On the other hand, permanent gullies are very large and cannot be smoothed by regular tillage (Blanco and Lal 2008). Gullies of various size and form develop by the gradual deepening of rills. A number of forms may be distinguished in gully erosion. The first form includes gully with a depth between 30 cm and 2–3 m. In this form,

Fig. 3.5 Gully erosion (Photo Courtesy of USDA-NRCS)

typical wash prevails with a marked backward or retrograde erosion and vertical or depth erosion. Gullies have larger dimensions and their development is more complicated. Besides retrograde and vertical erosion, lateral erosion also appears here, together with accessory landslide, soil flow, and other phenomena. Gullies may grow into gorges and canyons in high altitudes and very steep slopes. Gullies may be flat, narrow, broad, and round. Flat forms occur mostly on shallow soil or in connection with a specific lithic structure of the slope. In this form, characterized by a broad V-section, lateral erosion prevails over vertical erosion. Narrow, acute forms are created with a narrow V-section, the breadth of the gully usually being equal to its depth or smaller. Broad gullies have a wide bottom and are U shaped. Here, lateral erosion prevails over depth erosion. Active gullies maintain steep or even perpendicular sides. It happens frequently that recent forms replace older forms so that their origin and age cannot be assessed from superficial observation. The main feature of gully erosion is the volume and velocity of water at the lowest level. The energy of flowing water increases its cutting and smashing power and often results in bank erosion.

3.3.5 *Tillage Erosion*

The Soil Science Society of America defines tillage as the mechanical manipulation of the soil for any purpose (SSSA 2008). The objectives of tillage include seedbed preparation, weed control, and incorporation of agricultural amendments. Sometimes tillage becomes intensive and continuous, drastically alters soil functions, and causes soil erosion. Blanco and Lal (2008) emphasize the significance of a new type of erosion "tillage erosion," which refers to the gradual soil translocation or displacement downhill caused by tillage operations (Lindstrom et al. 1990). The mechanical manipulation of tillage implements transports soil and does not involve the water action. The net soil translocation by tillage is expressed in units of volume, mass, or depth per unit of tillage width. In recent years, particularly with the advent of mechanized agriculture, tillage erosion has become an important component of total soil erosion in hilly croplands. 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. 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).

3.3.6 *Landslide/Landslip/Mudflow*

The downward and outward movement of a large block of soil and regolith caused by gravity are called landslides (Fig. 3.6) and landslips. Landslides are deep-seated mass movement, and soil slip is a shallow and rapid sliding or flowing movement of the soil. There are different forms of landslides, including mudflows, mudslides, debris flows, rock falls, and rockslides. Slides move in contact with the underlying surface. Flows are plastic or liquid movements in which land mass breaks up in water and flows during movement. Landslides are caused by unstable geological conditions, steep slopes, intense rainfall, weak soils, earthquakes, and human-induced changes of landforms. Human-induced causes are excavation, loading, deforestation, irrigation, mining, vibrations, and water impoundment. The volume of soil detached by landslides depends on the geology and the mechanism of landslide. For example, landslides initiated by rainfall are smaller, while those initiated by earthquakes are very large. There are two categories of landslide: (1) rapidly moving and (2) slow moving. Rapidly moving landslides (debris flows and earth flows) present the greatest risk to human life.

3.3.7 *Riverbank/Stream Bank Erosion*

Stream/riverbank erosion occurs due to bank scour and mass failure. The direct removal of bank materials by the physical action of flowing water is called bank scour.

Fig. 3.6 Landslide on a hillslope (Photo courtesy of Dr. Animesh Biswas)

It is often dominant in smaller streams and the upper reaches of larger streams and rivers. Mass failure occurs when large chunks of bank material become unstable and topple into the stream or river (Fig. 3.7). Riverbank erosion can be accelerated by lowering streambed, inundation of bank soils followed by rapid drops in water flow, saturation of banks from off-stream sources, removal of protective vegetation from stream banks, poor drainage, readily erodible material within the bank profile, wave action generated by boats, excessive sand and gravel extraction, and intense rainfall.

3.3.8 *Effects of Water Erosion*

Soil erosion may be either harmless (benignant) or harmful (malignant). In well-vegetated forestlands, in pasturelands, and in level and mulched croplands, natural erosion is low, gradual, and harmless. Here, the rate of soil loss is less than the rate of soil formation. The estimated average soil erosion tolerance level (*T*) used in soil and water conservation planning in the USA is 11 Mg ha^{-1} $year^{-1}$. 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 (Blanco and Lal 2008). A survey of soil loss at 70 sites throughout Western Australia completed as a part of a National Reconnaissance Survey has shown that erosion is occurring at an unsustainable rate. Results indicated that about 10 % of the sites had soil losses in excess of 20 t ha^{-1} $year^{-1}$, about 30 % of the sites had soil losses of

Fig. 3.7 Riverbank erosion (Photo courtesy of Stephen Hallett of Soil-Net, Cranfield University, UK)

5–20 t ha^{-1} year^{-1}, about 40 % of the sites had soil losses of 1–5 t ha^{-1} year^{-1}, and about 25 % of the sites had negligible soil loss (less than 1 t ha^{-1} year^{-1}) (Foster et al. 2006). Erosion may remove the entire soil profile in some situation in lithosols (shallow soils on steep slopes). Sediment yields from urban earthworks can be extremely high, sometimes reaching values of 50,000 t ha^{-1} year^{-1} (Novotny and Chesters 1981). Figures of 100–120 t ha^{-1} year^{-1} are reported by Williamson (1993) from New Zealand areas undergoing construction. In the Auckland region, a measured loss of 66 t ha^{-1} year^{-1} was obtained.

It is estimated that about 1,960 M ha 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). On-site effects of water erosion include soil loss and loss of organic matter as well as nutrients (10 mm topsoil loss equals to 350 kg ha^{-1} N, 90 kg ha^{-1} P, and 1,000 kg ha^{-1} K; Hicks and Anthony 2001). Other on-site effects are surface sealing and soil compaction, exposure of roots, deformation of terrain, difficulty in tillage operations, exposure of subsoil, reduction in growth and yield of crops, loss of growing crops, decline in soil quality, and reduced capability of ecosystem functions. Erosion can cause yield reductions of 30–90 % in some root-restrictive shallow lands of West Africa (Mbagwu et al. 1984; Lal 1987). Yield reductions of 20–40 % have been measured for row crops in Ohio (Fahnestock et al. 1995) and elsewhere in Midwest USA (Schumacher et al. 1994). The productivity of some lands in Africa has declined by 50 % as a result of soil erosion and desertification (Dregne 1990). Yield reduction in Africa due to past soil erosion may range from 2 to 40 %, with a mean loss of 8.2 % for the continent (Lal 1995).

Off-site effects of soil erosion are not always easily noticed. Eroded materials are carried to distant places and get deposited there in water bodies, including lakes, streams, rivers, and on agricultural land, roads and highways, homesteads, and other installations. Off-site effects of water erosion include burrowing of crops and households, siltation on lands, sedimentation of reservoirs, floods, eutrophication of water bodies, pollution of land and water, and lowering of water quality. Sediment which reaches streams or watercourses can clog drainage ditches and stream channels, cover fish spawning grounds, and reduce downstream water quality. Pesticides and fertilizers, frequently transported along with the eroding soil, can contaminate or pollute downstream water sources and recreational areas. Because of the potential seriousness of some of the off-site impacts, the control of "non-point" pollution from agricultural land has become of increasing importance.

3.3.9 Soil Loss Equations (USLE and RUSLE)

An equation for the estimation of the amount of soil loss due to water erosion was proposed by Wischmeier and Smith (1965). They estimated soil loss due to sheet and rill erosion from farm-level experimental data on various soils at many different locations in the USA using the same standard conditions. Erosion plots were 22.6 m long on 9 % slopes and were subjected to the same soil management practices. This equation has been applied satisfactorily in many other areas than USA and, through revisions of the factors included in the equation (Wischmeier and Smith 1978), has become universally accepted. It is known as the Universal Soil Loss Equation (USLE). The USLE was further revised to give the Revised Universal Soil Loss Equation (RUSLE). In 1987 and early 1988, Porter wrote a computer program that would do the computations for RUSLE. It was built using the existing R factor database, supplemented with a climate database needed to estimate residue decomposition (monthly temperature and precipitation). It used a time varying soil erodibility value, based upon the standard USLE K value and temperature, the standard LS value for the USLE, a subfactor approach based upon the work of Laflen et al. (1985, 1990), which was built in part from the subfactor approach of Wischmeier. P factors were based upon the existing approaches used in the USLE (Laflen and Moldenhauer 2003). The RUSLE computes sheet and rill erosion from rainfall and the associated runoff on landscape scale. It incorporates data from rangeland and other research sites in the USA to significantly improve erosion estimates on untilled lands. It can be used to compute soil loss on areas where significant overland flow occurs but is not designed for lands where no overland flow occurs, such as undisturbed forestlands. The soil loss is an average erosion rate for the landscape profile. RUSLE uses the same factorial approach employed by the USLE:

$$A = R \times K \times LS \times C \times P$$

A – the potential long-term average annual soil loss in tons per acre per year ($\times 2.24$ Mg ha^{-1} year^{-1}). This is the amount that is compared to the "tolerable soil loss" limits.

R – the rainfall and runoff factor. The greater the intensity and duration of the rainfall, the higher the erosion potential.

K – the soil erodibility factor. *K* is a measure of the susceptibility of soil particles to detachment and transport by rainfall and runoff. Texture is the principal factor affecting *K*, but structure, organic matter, and permeability also contribute.

LS – the slope length and steepness factor. The LS factor represents a ratio of soil loss under given conditions. The steeper and longer the slope is, the higher is the risk for erosion. This is a very important factor in the overall erosion rate.

C – the crop management factor. It is used to determine the relative effectiveness of soil and crop management systems in terms of preventing soil loss. The *C* factor is a ratio comparing the soil loss from land under a specific crop and management system to the corresponding loss from continuously fallow and tilled land. The crop grown, type and timing of tillage, the use of winter cover, and the application of solid manure will all impact on the *C* factor.

P – the support practices factor. The *P* factor compares the soil losses from up- and downslope farming to losses that result from practices such as cross-slope cultivation, contour farming, and strip cropping.

In RUSLE, the factors have been updated with recent information, and new factor relationships have been derived based on modern erosion theory and data. Major changes to the USLE incorporated into RUSLE include:

R factor: new and improved isoerodent maps and erodibility index distributions for some areas

K factor: time-variant soil erodibility which reflects freeze – thaw in some geographic areas

LS factor: new equations to account for slope length and steepness

C factor: additional subfactors for evaluating the cover and management factor for cropland and rangeland

P factor: new conservation practice values for cropland and rangeland

3.3.9.1 Rainfall Erosivity Factor (*R*)

Generally, local variations in rainfall erosivity (+5 %) can be represented with a single *R* value. *R* values can be calculated for specific locations from rainfall intensity data. However, this is a very time- and labor-intensive process requiring erodibility index calculations for each storm event greater than 0.5 in. for each rain gauge over a period of years. “Equivalent” *R* values have been developed for many areas in the USA. The effect of the distribution of *R* over the year is incorporated into the model in the computation of the *C* factor.

3.3.9.2 Soil Erodibility Factor (*K*)

The *K* factor represents both susceptibility of soil to erosion and the amount and rate of runoff. Soil texture, organic matter, structure, and permeability determine the erodibility of a particular soil. *K* values for various soil types are given below.

Soil type	Erodibility	*K* value range
Fine-textured; high in clay	Low	0.05–0.15
Coarse-textured; sandy	Low	0.05–0.20
Medium-textured; loams	Moderate	0.25–0.45
High silt content	High	0.45–0.65

Although soil organic matter reduces erodibility, extrapolation of the *K* factor nomograph beyond an organic matter of 4 % is neither recommended by the NRCS nor allowed by RUSLE software. Soil structure affects erodibility by influencing detachment and infiltration. Permeability of the soil profile affects *K* because it affects runoff (Laflen and Moldenhauer 2003).

3.3.9.3 Slope Length and Steepness Factor (LS)

The LS factor represents the combined effects of slope length and steepness relative to a standard unit plot on the erodibility. Slopes of nonuniform steepness require dividing the slope into segments. Usually, five segments comprised of slope length and steepness are sufficient to define a nonuniform slope profile. There are different equations for calculating LS factor for different slope conditions.

3.3.9.4 Cover and Management Factor (*C*)

The *C* factor represents the effect of plants, soil cover, belowground biomass, and soil-disturbing activities on soil erosion. Both time-variant (cropping/rotation scenario) and time-invariant (average annual values) modules have been constructed.

The time-variant option is used when plant and/or soil conditions change enough to significantly affect erosion during the year, during a rotation cycle, or over an extended period. This option is typically applied to croplands; rangelands where cover changes significantly during the year such as from grazing, burning, or herbicide application. The time-invariant option is used where constant conditions can be assumed. The *C* factor depends on effective root mass in top 4″ of soil, percent canopy, average fall height (ft), surface roughness value (index of average micro-elevation: generally ranges from 0.3 to 1.5), percent ground cover (rock + litter, excluding plant basal cover), and surface cover function expressed as *B* value (the relative effectiveness of surface cover for reducing soil loss). The choice of *B* value is based on the ratio of rill/interrill erosion under bare soil conditions. Some typical *B* values are given below:

Field Conditions (*B* Value)

Flat and short slopes, where soil is resistant to erosion by flow, consolidated lands (e.g., pasture)	0.025
Moderate slopes and slope lengths with moderate disturbance	0.035
Steep and long slopes where soil is highly disturbed and where soil is susceptible to erosion by flow	0.045
Range lands, where runoff tends to be low and affected by cover	0.045
Long-term no-till cropping, especially where no-till significantly reduces runoff	0.050

3.3.10 Soil Loss Tolerance Value (T Value)

The soil loss tolerance value has been defined as an indication of how much erosion should be tolerated. The *T* value is the maximum soil erosion loss that does not cause significant loss in productivity. It depends on soil characteristics. For example, shallow soils over hard bedrock have small *T* values. More erosion loss can be tolerated for thick permeable soils on permeable unconsolidated parent materials. The *T* value is the maximum average annual soil loss that allows continuous cropping and maintains soil productivity without requiring additional management inputs. Many soils that have developed from thick sediments of loess are agriculturally productive. Where subsoils have physical properties unsuitable for rooting, erosion results in reductions in soil productivity that cannot be overcome with only fertilizer application. Such soils have low tolerance levels (small *T* value). The average annual rate of soil erosion on cropland in the USA is 11 t ha^{-1}. *T* values in the USA range from 2 to 11 t ha^{-1} (Foth 1990).

3.4 Control of Water Erosion

Control of soil erosion by water is based on the following principles:

- Reducing raindrop impact: This can be achieved by providing a cover on the soil during the rainy season. Dense forest canopy, close-growing crops such as cover crops and mulches on the bare or cropped soils can provide necessary protection against raindrop impact.
- Stabilizing soil aggregates: Stable soil aggregates are obtained in soils supplied with sufficient organic matter. Aggregation improves porosity and infiltration and reduces runoff.
- Increasing infiltration and reducing runoff: Infiltration can be increased by mulching and by modification of the slope. Organic mulches soak water and allow water more time to infiltrate. Level lands have more infiltration capacity than sloping soils.
- Reducing velocity of runoff: Velocity of runoff can be reduced by modifying the degree and length of slope through terracing and contouring. Contour cropping, strip cropping, and contour strip cropping effectively reduce runoff velocity. When velocity of runoff is reduced, rate of infiltration increases.
- Minimum disturbance of soil: Tillage makes the soil more erodible. Conservation tillage systems, including no-tillage, minimum tillage, and subsoil tillage are efficient soil conservation practices.
- Preventing concentration of runoff water in channels: Leveling previously developed rills, growing crops closely, and keeping crop residues in field prevent concentration of runoff water.
- Carrying runoff water safely out of field. Runoff water can be driven safely out of the field by grassed waterways.

- Integrating erosion control measures: Usually, no one method alone is sufficient for the control of soil erosion. For example, integrating mulching with no-tillage can effectively reduce erosion.
- Regular maintenance of erosion control measures. Practices for erosion control need to be maintained regularly. Terraces may need mending and barriers may need reconstruction.

3.4.1 Amendments

Addition of manures and composts favors structure formation; increases aggregate stability, porosity, and infiltration; and thus reduces runoff. 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 may be an effective strategy for reducing soil erosion. However, indiscriminate use of manure may have detrimental impacts on water quality as well.

Farmers in some parts of the USA and other areas are increasingly using polyacrylamide (PAM) in irrigated soils. The PAM molecules stabilize the aggregates and reduce soil particle detachment. 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). Some steeply sloping heavily irrigated soils may need as high as 20 2–4 kg ha^{-1}. It reduces soil erosion by about 1×10^6 Mg annually in Western and Northern USA (Sirjacobs et al. 2000). Its 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.

3.4.2 Cover Crops

According to the Soil Science Society of America, 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). Cover crops can be used as green manure crops. Cover crops are used for soil conservation from ancient times in Greece, Rome, China, and other regions (Magdoff 1992). Cover crops are presently used 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. Cover crops offer the following benefits: (i) protecting soil against erosion, (ii) improving soil properties, (iii) enhancing soil fertility, (iv) suppressing weeds, (v) fixing N, (vi) increasing soil organic matter content, (vii) increasing crop

Fig. 3.8 Cover crops on a field in Black Hawk County, Iowa (Photo courtesy of USDA-NRCS)

yields, (viii) recycling nutrients, (ix) preventing leaching of nutrients, and (x) improving water quality (Blanco and Lal 2008). Cover crops are mainly grown between the cropping seasons. They can also be grown as rotational crops and companions to main crops. Cover crops may include suitable grasses and legumes. When legumes are grown as cover crops, biological nitrogen fixation may benefit the succeeding crop. Field legumes may fix 200–300 kg N ha^{-1} $year^{-1}$. 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. Establishing cover crops is one of the top conservation practices for reducing runoff and soil erosion from agricultural soils (Fig. 3.8).

High seed rates and fertilizers and manures are used to obtain a dense stand of cover crops. Incorporation of cover crop as a green manure is recommended prior to blooming. Green manuring with leguminous cover crops while they are tender improves decomposition, increases biological activity, causes a rapid nutrient release, and improves supply of nitrogen for the next crop. Cover crops must be incorporated into the soil several weeks prior to planting the main crops to minimize risks of drought stress in semiarid and arid regions and to avoid competition for nutrients by decomposing microorganisms. Retaining cover crops as mulch is more benefitting than plowing under in soils where the erosion rate is high. Cover crop mulch on the soil surface increases soil organic matter content and suppresses weeds in addition to protection against erosion (Blanco and Lal 2008).

3.4.3 Conservation Tillage

Tillage refers to the mechanical manipulation of soil for cultivation of crops. Tillage involves a series of mechanical operations including plowing and harrowing to prepare a good seedbed. The main objectives of tillage are inverting the soil, incorporating crop residues and amendments, controlling weeds, and harvesting crops. Tillage systems may be grouped into conventional tillage and conservation tillage.

Conventional tillage is any tillage system that inverts the soil and alters the natural soil structure. It includes loosening soil, incorporating residues, preparing a seedbed, controlling weeds, and mixing lime, fertilizer, and other chemicals. It primarily involves the moldboard plowing. The introduction of moldboard plow increased food supply particularly in developed countries. But intensive plowing has caused soil erosion, depleted soil fertility, and reduced biological activities. 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. Because of its adverse impacts, use of moldboard plow has decreased since the 1970s, especially in the USA, Canada, Brazil, Argentina, Australia, etc.

Conservation tillage is any system that reduces the number of tillage operations, reduces the area of tilling in the field, and maintains residue cover on the soil surface. The Soil Science Society of America (SSSA 2008) defines conservation tillage as a tillage system that leaves at least 30 % of residue cover on the soil surface. When combined with prudent management of crop residues, crop rotations, and cover crops, conservation tillage is a useful technology for protecting soil and sustaining crop production (Blanco and Lal 2008). Conservation agriculture occupies about 100 M ha of land worldwide (Derpsch 2005). Conservation tillage includes no-till and reduced or minimum tillage systems such as mulch tillage, strip tillage, and ridge tillage. Cropping with no-tillage or limited tillage is not as popular as with tillage. But for negative impacts of conventional tillage, including exposure of the soil surface to wind and water erosion, and loss of soil organic matter through oxidation, conservation tillage is gaining popularity. These management strategies have proved effective for controlling soil erosion and improving soil quality.

In no-tillage, crops are planted directly in the residues of the previous crop with no prior tillage (Fig. 3.9). For row crops, a slit is made in the soil in which the seed is sown. Minimum tillage involves the minimum manipulation of soil. It is actually a localized tillage. For example, strawberry in Fig. 3.10 has been grown by tilling only the soil of the rows. Minimum and no-tillage leave more residues on the soil surface than conventional tillage, resulting in enhanced infiltration and reduced runoff and soil erosion for which they are called conservation tillage. A strip 30–45 cm wide is tilled in the row between undisturbed spaces during strip tillage. Strip-till is less effective than no-till and subsoil systems, because bare soil exposed in the tilled strip is susceptible to erosion. It can be made effective by covering the exposed part with organic residues. Conservation tillage systems have some disadvantages too. Yield of crop tends to be lower and weed infestation is a major problem.

Fig. 3.9 Young soybean seedlings in a no-till field (Photo courtesy of USDA-NRCS)

Fig. 3.10 Strawberries grown through reduced tillage (Photo courtesy of USDA-NRCS)

Fig. 3.11 Soybean mulched with corn straw (Photo courtesy of USDA-NRCS)

3.4.4 Mulching

A cover spread or laid over the surface to protect soil is called a mulch. Organic mulches are efficient soil conservation measures (Fig. 3.11). Organic mulches include compost, composted manure, grass clippings, newspaper, straw, and shredded leaves. They are natural and cheap, and along with protection, they improve soil fertility. Mulching has multiple advantages. It reduces the impact of solar radiation and raindrops. It protects soil aggregates from detachment by raindrops. It reduces evaporation and loss of soil moisture; it increases infiltration and reduces amount and velocity of runoff. It decreases surface sealing, crusting, and compaction. Stubble mulching is frequently recommended for reducing runoff and erosion. Residues of wheat or stalk of corn of the previous crop are retained during tilling the land for next crop without turning them under. Stubble mulch is a very efficient protector of wind erosion. Organic mulches release polysaccharides, polyuronides, and other cementing agents which improve soil structure. Mulched plants have more roots than plants that are not mulched.

Inorganic mulches also provide many benefits to the landscape. Materials that can be used as inorganic mulches are crushed gravel and granite, river rock or small

Fig. 3.12 Contour cropping (Photo courtesy of USDA-NRCS)

stones, lava or granite rock, decorative and colored stones, sand, crushed brick, crushed graded recycled glass, landscape fabric sometimes referred to as geotextiles, plastic mulch, and aluminized mulch.

3.4.5 Contour Cropping

Plowing and planting crop in the contour that is across the slope is called contour cropping or contour farming (Fig. 3.12). Contours are arbitrary lines drawn perpendicular to the direction of slope. So, contour farming is a cross-slope farming system. Contours reduce velocity of runoff, give accumulated water more time to infiltrate, and deposit detached soil particles along the contour lines. It retains sediments in the field. In contour farming, ridges and furrows are formed by tillage, planting, and other farming operations to change the direction of runoff from directly downslope to around the hillslope. Contour farming is most effective on slopes between 2 and 10 %. Contour farming is not well suited to rolling topography having a high degree of slope irregularity.

Several factors influence the effectiveness of contour farming to reduce soil erosion. They are rainfall intensity, slope steepness, soil properties, ridge height, cover and roughness, and the critical slope length. Cover, roughness, and ridge height can be influenced by management. Spacing of contour lines is chosen on the basis of slope, soil, rainfall, and crop type. Annual and perennial crops are planted in the ridges or furrows of the contours. Contour farming can be combined with strip cropping.

Fig. 3.13 Strip cropping (Photo courtesy of USDA-NRCS)

3.4.6 *Strip Cropping*

In strip cropping, two or more crops are grown in alternate strips (Fig. 3.13). Crops of different strips vary in their root/shoot characteristics and cultural requirements. Crop strips break sloping landscapes in wide segments with diverse vegetative cover which intercepts runoff and promotes water infiltration, thereby reducing runoff and soil erosion. Sod-forming crops may be alternated with cereals, legumes with non-legumes, and root crops with vegetables. Strip cropping gives yields as good as monocropping. 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 be successfully combined with contour farming.

3.4.7 *Contour Strip Cropping*

Contour strip cropping is planting row crops in strips on the contour (Fig. 3.14). It is more efficient in erosion control than contour farming and strip cropping alone because of the plant and crop diversity. The grass, legumes, or small grains used in strips slow runoff and trap sediments leaving row crops. Permanent grass/legume

Fig. 3.14 Contour strip cropping (Photo courtesy of USDA-NRCS)

strips must be maintained between strips in soils with severe erosion. These strips can be used as traffic lanes for cultural operations (Blanco and Lal 2008).

3.4.8 Sloping Agricultural Land Technology (SALT)

SALT is a technology package of soil conservation and food production that integrates several soil conservation measures (Tacio 1993). It involves planting field crops and perennial crops in bands 3–5 m wide between double rows of nitrogen-fixing shrubs and trees planted along the contour. Thus, it is a modification of the contour strip cropping, but it may be practiced in land of slope more than 10 %. Field crops include legumes, cereals, and vegetables while the main perennial crops are cacao, coffee, banana, citrus, and fruit trees (MBRLC 1988). The nitrogen-fixing trees are thickly planted in double rows to make hedgerows (Fig. 3.15). When a hedge is 1.5–2 m tall, it is cut down to about 75 cm, and the cuttings are placed in alleyways to serve as organic fertilizers. SALT establishes a diversified stable ecosystem. SALT was developed on a marginal site in the Philippines by the Mindanao Baptist Rural Life Center (MBRLC) in 1971. There are ten steps in establishing a SALT farm:

Step 1 Making the A-frame: A-frame is a wooden frame of two legs in the shape of English alphabet “A” having a water or sprit level on the middle arm.

Step 2 Drawing contour lines: Contour lines are drawn by joining the points of equal elevations selected by the A-frame in a slope.

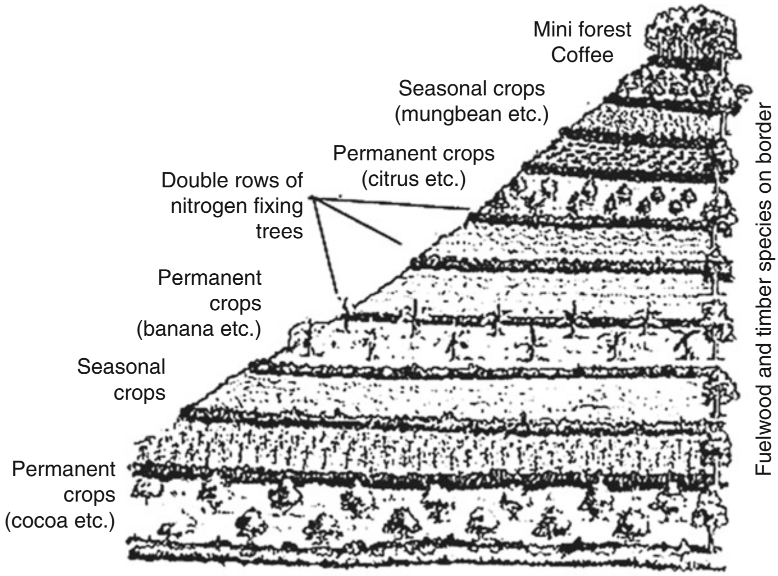

Fig. 3.15 Outline of a SALT farm (Sketch courtesy of Agnet)

Step 3 Cultivating the contour lines: One-meter strips along the contour lines should be plowed and harrowed to prepare for planting.

Step 4 Planting seeds of different nitrogen-fixing trees and shrubs: Along each prepared contour line, two furrows should be laid out. Leguminous tree seeds (or shrubs) are sown in double rows 12 cm apart. Tree or shrub seedlings will form dense hedgerows which will intercept runoff and trap soil materials. Suitable hedgerow species are *Flemingia macrophylla* (syn. *congesta*), *Desmodium rensonii*, *Calliandra calothyrsus*, *Gliricidia sepium*, *Leucaena diversifolia*, and *L. leucocephala*, etc.

Step 5 Growing crops between hedgerows: Crops are grown in the space between the hedgerows. It is called a strip or alley. Cultivation is done on alternate strips.

Step 6 Planting permanent crops: Permanent crops such as coffee, cacao, banana, citrus, and others of the same height may be planted in cleared spots of hedgerows. Weeding is employed until the hedgerows are large enough to hold the soil in place. Permanent crops are planted in every third strip. Tall crops should be planted at the bottom of the farm, while the short ones are planted at the top.

Step 7 Planting short-term crops: Short- and medium-term cash crops (pineapple, ginger, sweet potato, peanut, mung bean, melon, sorghum, corn, upland rice, etc.) should be planted between the strips of permanent crops.

Step 8 Trimming of nitrogen-fixing trees: Every 30–45 days, the growing hedgerows are cut to a height of 1.0–1.5 m from the ground. The prunings are piled on the soil around the crops as organic fertilizer.

Step 9 Practicing crop rotation: A good way of rotating is to plant cereals such as corn or upland rice, tubers, and other crops on strips where legumes were planted previously.

Step 10 Building green terraces: To enrich the soil and effectively control erosion, organic materials such as straw, stalks, twigs, branches and leaves, and also rocks and stones are piled at the base of the rows of nitrogen-fixing trees. By the passage of time, permanent effective terraces will be formed.

SALT has been applied in many other countries with variable success. However, it may be an alternative to shifting cultivation in degraded hilly lands.

3.4.9 Agroforestry

Agroforestry is a system of growing agricultural or horticultural crops or/and rearing livestock along with trees simultaneously or sequentially in the same piece of land. The objectives of agroforestry include conserving soil, recycling nutrients, and enhancing crop yields, while producing fuelwood, fodder, grain, fruit, and timber (Nair 1989). It involves the integration of trees, plants, and animals in conservative, long-term, productive systems. The positive interactions among all these components are exploited in carefully designed sustainable agroforestry systems (Sanchez 1995) by (i) multiple use of land; (ii) improved utilization of land, labor, and resources; (iii) protection and improvement of soil by reducing erosion and providing soil organic matter; (iv) production of diverse food crops such as fruits, nuts, grains, and seeds; (v) production of feed for farm animals; (vi) long-term production of tree products; and (vii) enhanced productivity and net economic returns. In agroforestry trees are used in these ways: (i) individual trees in home gardens, around houses, paths, and public places; (ii) dispersed trees in cropland and pastures; (iii) rows of trees with crops between; (iv) strips of trees along contours or waterways; (v) living fences and borderlines; and (vi) windbreaks. There are two functionally different types of agroforestry systems: simultaneous and sequential (ICRAF 1994). In simultaneous agroforestry system, the tree and the crop components grow at the same time and in close enough proximity for interactions to occur. Examples of this type are alley cropping, contour buffering, border planting, and different silvopastoral systems. In sequential agroforestry systems, the maximum growth rates of the crop and the tree components occur at different times even though both components may have been planted at the same time and are in close proximity. Examples of this type are shifting cultivation, improved fallows, taungya, and some multistrata systems.

3.4.10 Alley Cropping

In alley cropping, crops (grains, forages, vegetables, etc.) are grown between tree rows spaced widely enough to accommodate the mature size of the trees without

Fig. 3.16 Alley cropping of cowpea with Leucaena (Photo courtesy of IITA)

interfering for light and moisture with the crops between the rows (Fig. 3.16). When light-demanding crops like corn (maize) will be grown, the alleyways need to be wide enough to let in plenty of light even when the trees have matured. Alternatively, the cropping sequence can be planned to change as the trees grow. For instance, soybeans or corn could be grown when the trees are very small; as the tree canopy closes, forages could be harvested for hay; and finally, when the trees are fully grown and the ground is more shaded, grazing livestock, or shade-tolerant crops like mushrooms or ornamental ferns could occupy the alleyways. For alley cropping, generally fast-growing multipurpose leguminous tree species including white lead tree (*Leucaena leucocephala*), Kassod tree (*Senna siamea*), Erythrina (*Erythrina poeppigiana*), and Gliricidia (*Gliricidia sepium*) are selected. Between these tree rows, corn (maize), rice, soybean, etc. are grown as alley crops.

3.4.11 Buffer Strips

Buffer strips are zones of permanent vegetation – trees, shrubs, and grasses – used for different purposes including reduction of erosion. Buffers reduce runoff by obstructing its way and cutting downslopes, filter sediments, and remove sediment-borne chemicals and dissolved nutrients and agrochemicals. Buffer strips are generally established between agricultural lands and streams, rivers, and lakes. When placed perpendicular to the direction of water flow, buffers are effective measures for reducing sediment fluxes. Buffers are commonly used in sloping lands of developing regions where access to heavy equipment and construction of mechanical

Fig. 3.17 Grassed waterway (Photo courtesy of USDA-NRCS)

structures (terraces) can be unachievable. Benefits of buffers include yield of good quality water, enhanced agricultural production, secured wildlife habitat, and desired landscape aesthetics. Buffers can trap >70 % of sediments and >50 % of nutrients depending on the plant species, management, and climate (Blanco and Lal 2008). Buffers are multifunctional systems. Above the surface, buffers reduce the runoff velocity and trap sediments and nutrients, and 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.

3.4.12 Grassed Waterways

Grassed waterways are natural or constructed channels established at an appropriate place over the field for safe transport of concentrated water at a reduced velocity using adequate grass cover (Fig. 3.17). They are generally broad and shallow drainways to transport surface water across farmland without causing soil erosion. Grassed waterways are used as outlets to prevent rill and gully formation. The grass cover slows the flow of water and minimizes channel erosion. Efficient grassed waterways can transport large water flows downslope without causing any harm. Grassed waterways also act as diversion channels. Grasses trap suspended sediments and absorb dissolved nutrients. It also traps dissolved nitrates, phosphates, herbicides, and pesticides and improves water quality of adjacent reservoirs.

3.4.13 Terracing

Terracing refers to the building of a mechanical structure, a channel and a bank or an earthen ridge or a stonewall on the land to reduce steepness of slope and divide the slope into short gently sloping sections (Morgan 1986). Terraces are created to encourage infiltration, to intercept surface runoff, or divert toward a predetermined and protected safe outlet at a controlled velocity to avoid soil erosion (USDA Soil Conservation Service 1980; FAO 2000). The critical runoff velocity, at which soil particles that have been detached from soil aggregates begin to be transported over the surface, is 5 m s^{-1} in sandy soils and 8 m s^{-1} in clay soils (Rufino 1989; FAO 2000). Terracing reduces runoff velocity below this threshold values. It is one of the oldest means of saving soil and water. Moreover, it is the most widely used soil conservation practice throughout the world (Hanway and Laflen 1974; Mountjoy and Gliessman 1988; Sutikto and Chikamori 1993; Skinner and Porter 1995; Drechsler and Settele 2001; Bokhtiar et al. 2001; Kasai et al. 2001).

3.4.13.1 Types of Terraces

Terraces can be naturally formed upslope contour hedgerows (Poudel et al. 1999), vegetative filter strips (Stark et al. 1999), and grass barriers (Aase and Pikul 1995; Walle and Sims 1999). Man-made terraces can be classified on different criteria. They can be classified according to (1) their main function, (2) the construction process, (3) the size of the terrace base, and 4) the shape.

1. Main Function of the Terrace
    (a) Retention terraces, also called absorption or level terraces (Morgan 1986): these are designed to accumulate and retain runoff in the terrace channel so that it will eventually infiltrate and the sediment accumulates. These terraces are recommended for low rainfall areas, permeable soils, and for land of less than 8 % slope. They are normally broad-based terraces (FAO 2000).
    (b) Graded or diversion terraces: these are sloping terraces, designed to intercept or divert runoff into protected waterways. These terraces are recommended for high rainfall regions, for slightly or moderately permeable soils, and for slopes of between 8 and 20 % (FAO 2000).
2. Construction Process
    (a) Channel or Nichols terrace: these terraces are constructed by excavating soil from the upper side only to form a channel and depositing it downhill to form a bank (Morgan 1986). They are recommended for slopes up to 20 %. They are used in high rainfall regions and in soils of low or medium permeability (FAO 2000).

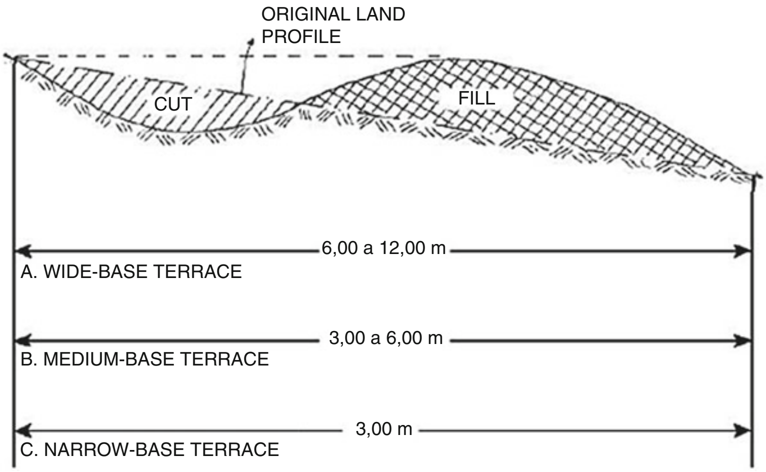

Fig 3.18 Types of terraces on size of the terrace base (FAO 2000)

(b) Ridge or Mangum terrace: a long, low ridge of earth with gently sloping sides and a shallow channel along the upper side. These terraces are constructed by excavating the soil from both sides of the embankment (Morgan 1986). They are recommended for slopes less than 10 %, for low rainfall regions, and for permeable soils (FAO 2000). Ridge terraces control erosion by diverting surface runoff across the slope instead of permitting it to flow uninterrupted down the slope.

3. Size of the Terrace Base (Fig. 3.18)

   (a) Narrow-base terraces: where soil movement is limited to about 3 m
   (b) Medium-base terraces: where soil movement is 3–6 m
   (c) Wide-base or broad-based terraces: where soil is moved more than 6 m, but normally less than 12 m

4. Terrace Shape
   According to their shapes, terraces could be of the following two types (Bertolini et al. 1989).

   (a) Normal terrace: Normal terrace consists of a ridge or bank and a channel, which may be constructed on a gradient or level. This type of terrace is normally used in areas where the slope is less than 20 % (FAO 2000). These terraces mostly include broad-based terraces.
   (b) Bench terraces: These terraces form a series of level or nearly level strips of earth and a steep or vertical downhill face, constructed on or nearly on the contour (Fig. 3.19). Bench terraces are constructed by cutting and filling and are used in land with slopes in excess of 20 %. The bench terrace is perhaps one of the oldest forms of terraces. All other types of terraces have been derived from this terrace type.

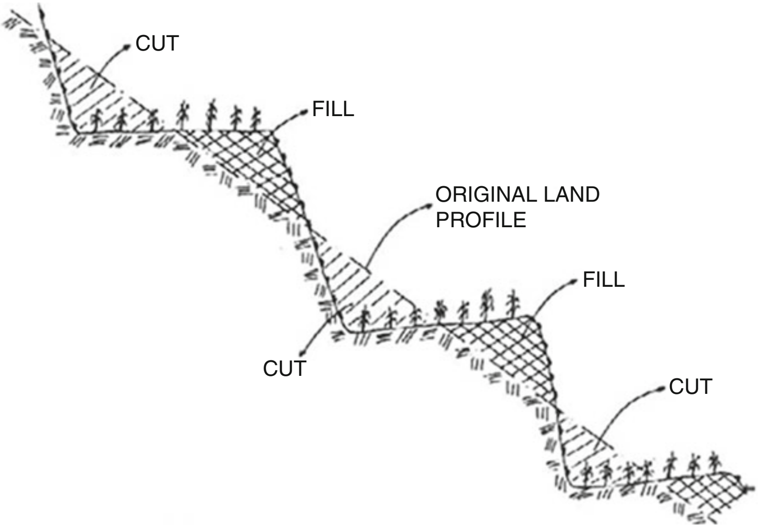

Fig. 3.19 A sketch of a typical bench terrace (FAO 2000)

The three types that are used most are bench terraces, contour terraces, and parallel terraces (Keirle 2002; NRCS 2004), although this subdivision mixes different criteria.

ICIMOD (1998) mentions the following limitations of terracing: the disturbance of the soil strata and considerable decline in soil fertility in the first several years, considerable soil loss during construction and in the first 2 years, and need of tremendous labor and investment for construction and maintenance. Further, they are not always stable in many cases and not suitable for sandy and coarse soils and on very steep land. Soil loss and nutrient leaching from bench terraces affect about 25 % of the marginal land.

Study Questions

1. Explain geological and accelerated erosion. What are the causes of accelerated soil erosion? Discuss the on-site and off-site effects of soil erosion caused by water.
2. Give an account of different types of soil erosion by water. Distinguish between interrill and rill erosion. Which type of water erosion will create the most severe management problems?
3. What are the principles of soil erosion control? Explain that for an efficient control of erosion you need integrated efforts.
4. Describe the advantages and disadvantages of mulching. Discuss the benefits of conservation tillage. How does contour cropping differ from strip cropping?
5. What are the objectives of terracing? Describe the different types of terraces with their suitability to different soil types.

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Chapter 4
Wind Erosion

For environmental and pedogenic reasons, soils of the arid and semiarid regions are usually dry, loose, low organic matter containing sandy soils susceptible to severe damage by wind erosion. Human activities such as deforestation, overgrazing, over exploitation of vegetation, soil and water resources have accelerated wind erosion many fold. Wind erosion removes the lighter and less dense soil constituents like organic matter, nutrients, clays, and silts. It significantly reduces soil productivity and crop yield. Many soils have been abandoned, and desertification has advanced in many places as an ultimate result of wind erosion. Effective soil conservation methods have been used successfully for the reduction of wind erosion, restoration of crop production, and rehabilitation of soils to agriculture. These measures include stabilizing soil, ridging and roughening soil, cover crops, residue management, mulching, mechanical barriers, and windbreak. Some innovative measures have been employed in stabilization and rehabilitation of dunes, such as straw checkerboard barriers in Michigan, USA, and Mauritania in West Africa, for example.

4.1 Causes of Wind Erosion

Wind erosion is a serious problem in the arid and semiarid regions where vegetation is sparse, rainfall is low, and temperature is high. Potential evaporation is higher than precipitation for most of the year, which causes depletion of soil moisture, organic matter, and structure. Storms are regular events there, and in dry warm season, strong winds uplift small soil particles and carry them to distant places. Ecosystems in arid and semiarid regions are fragile by nature and are sensitive to human disturbances. Under population pressure and socioeconomic backwardness, human actions cause stresses on all natural resources. Land mismanagement, overgrazing, overcutting for fuelwood and deforestation, and misuse of water resources have been responsible for the loss of natural vegetative cover and hence accelerated wind erosion. Comparative wind tunnel experiments have shown that wind erosion may

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Fig. 4.1 Severe wind erosion may lead to desertification (Photo courtesy of USDA-NRCS)

be accelerated more than a factor of 10 by cultivation, a factor of 1.14 by overgrazing, and a factor of 22.8 by overcutting (Dong et al. 1987; Hu et al. 1991; Liu et al. 1992). Yan (2000) estimated using ^{137}Cs measurements that in Gonghe Basin of Qinghai Province, China, the annual wind erosion rate of dry farmland is three times greater than that of adjacent grassland, and that during cultivation wind erosion may increase to a factor of 5–8 over the adjacent grassland. As a whole, the increased wind erosion caused by human factors on an average accounts for approximately 78 % of the total wind erosion (Liu et al. 1992; Wang and Wu 1999).

4.2 Effects of Wind Erosion

In the USA, wind erosion damages from 0.4 to 6 M ha of land annually, and about 2 M ha is moderately to severely damaged each year (Gregory and Borrelli 1986). Wind erosion not only removes soil but also damages crops, buildings, fences, and highways. Wind erosion removes the lighter, less dense soil constituents such as organic matter, clays, and silts. It causes loss of the most fertile part of the soil and significantly reduces soil productivity. If reversion measures are not taken and wind erosion continues for long, it may ultimately lead to desertification (Fig. 4.1).

Lyles (1975) estimated that top soil loss from wind erosion causes annual yield reductions of about 340,000 bushels of wheat and 545,000 bushels of grain sorghum on 0.5 M ha of sandy soils in southwestern Kansas, USA. Blowing soil impacting plants can also reduce seedling survival and growth, depress crop yields, lower the marketability of vegetable crops, increase the susceptibility of plants to certain types of stress, including diseases, and contribute to transmission to some plant pathogens (Armbrust 1984; Michels et al. 1995). In the long run, the cost of wind erosion control practices can offset the cost of replanting a blown out crop. Some soil from damaged land enters suspension and becomes part of the atmospheric dust load. Dust obscures visibility and pollutes the air, impacts water quality, causes automobile accidents, fouls machinery, and imperils animal and human health (Skidmore 1988).

Areas susceptible to wind erosion around the world are in the USA and Canada in North America; in drier portions of Argentina, Bolivia, and Peru in South America; in both European and Asiatic parts of Russia; in China, India, and Pakistan and much of the Middle East in Asia; north and south of the equator in Africa; and in Australia (Skidmore 1986). It is estimated that the degraded area caused by wind erosion amounts to 5.05×10^6 km^2, accounting for 46.4 % of the global degraded land (UNEP and ISRIC 1990).

4.3 Processes of Wind Erosion

There are three typical processes of soil particle movement during wind erosion. These are saltation, suspension, and surface creep. These three processes of wind erosion occur simultaneously. Saltation causes other particles to move in suspension and surface creep. Neither creep nor suspension can occur without saltation. Lyles (1988) mentioned that the following processes are involved in wind erosion: initiation, transport (suspension, saltation, and surface creep), abrasion, sorting, and deposition of soil particles.

4.3.1 Saltation

In saltation, fine soil particles (0.1–0.5 mm in diameter) are rolled over the soil surface by direct wind pressure to some distance and then abruptly jump up vertically to a height of 20–30 cm. Lifted particles gain in velocity and then descend in an almost straight line at an angle 5–12° from the horizontal. The horizontal distance traveled by a particle is four to five times the height of its jump. On striking the surface, the particles may rebound into the air or knock other particles into the air before coming to rest. Thus, saltation is a progression of particles of successive jumps (Fig. 4.2). As the saltating particles crash into the surface, they splash up more particles that also bounce across the surface. This bombardment of the surface causes an avalanching effect that spreads out in a fan shape, with more and more soil particles being mobilized downwind. Between 50 and 75 % of the soil is carried by saltation.

Bagnold (1941) suggested that saltating particles rebound from a surface at 90° and follow a characteristic path length. It is now shown from wind tunnel experiments using high-speed cine photography demonstrated that saltating grains impacted the surface at an angle with the horizontal of about 10° and on average rebounded at 25°, although the

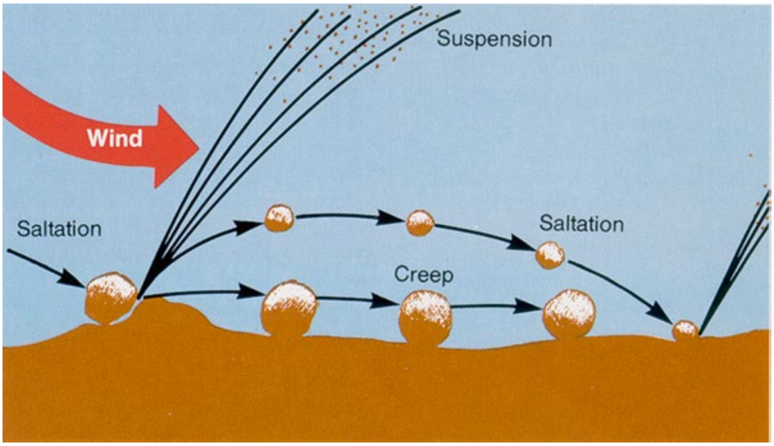

Fig. 4.2 Wind erosion processes (Image source http://www.weru.ksu.edu/new_weru/images/CreepSaltSusp.jpg)

latter depended on particle size and on the configuration and slope of the bed (Willetts and Rice 1989; McEwan et al. 1992; Rice et al. 1995). With non-cohesive sand-sized particulate surfaces, the saltating particles splashed up previously stationary grains.

4.3.2 Suspension

Suspension refers to the vertical uplift and horizontal transport of very small soil particles that are generally removed from the local source area. Suspended particles may end up on some meters or hundreds of kilometers downwind. They can range in size from about 2 to 100 μm, with mass median diameter of about 50 μm in an eroding field (Chepil 1957b; Gillette and Walker 1977). However, in long-distance transport, particles <20 μm in diameter predominate because the larger particles have significant sedimentation velocities (Gillette 1977). Some suspension-size particles are present in the soil, but most are created by abrasive breakdown during erosion. Because organic matter and some plant nutrients are usually associated with the finer soil fractions, suspension samples are enriched in such constituents compared with the bulk soil source. In the Great Plains, Hagen and Woodruff (1973) found that the average dust storm lasted 6.6 h and estimated the median dust concentration to be 4.83 mg m^{-3}. Suspension movements are easily noticed as dust storms.

4.3.3 Surface Creep

Soil particles or aggregates of 500–1,000 μm diameter are too large to be lifted up in normal erosive winds. They are pushed, rolled, and driven by the impacts of spinning particles in saltation. In high winds, the whole surface appears to be creeping slowly forward. The rippling of wind-blown sand has been attributed to unevenness in surface creep flow (Bagnold 1941). Surface creep constitutes 7–25 % of total transport (Bagnold 1941; Chepil 1945; Horikawa and Shen 1960). Creep appears nearly passive in the erosion process, but creep-sized aggregates may abrade into the size range of saltation and suspension and, thus, shift modes of transport. Creep aggregates seldom move far from their points of origin.

4.3.4 Abrasion

The percentage of erodible soil (<1,000 pm) in the surface layer is highly correlated with the mass of soil removable from that surface in wind tunnel tests (Chepil 1958). On long, erosion-susceptible fields, the total amount of soil that can be lost is usually several times the amount of erodible material initially present at the surface. Thus, resistance to abrasive breakdown of surface aggregates is important in wind erosion. The abrasion susceptibility of soil can be defined as the mass of material abraded from target aggregates per unit mass of impacting abrader (Lyles 1988).

4.3.5 Sorting

Unless surface-layer aggregates or particles are homogeneous in physical properties (size, shape, density), which is highly unlikely in agricultural soils, sorting will occur during erosion. Sorting here refers to the selective removal during erosion of aggregates or particles because various sizes move at different mass flow rates. The sorting process over time removes the finer, nutrient-enriched materials, leaving behind those that are coarser and less fertile (Lyles 1988).

4.4 Factors Affecting Wind Erosion

Several factors of wind erosion have been included in the wind erosion equation (WEQ), which is an erosion model designed to predict long-term average annual soil losses from a field having specific characteristics. The equation is shown as

$$E = f\left(IKCLV\right)$$

where

E = estimated average annual soil loss expressed in tons per acre per year
I = soil erodibility index
K = soil ridge roughness factor

C = climatic factor
L = equivalent unsheltered distance across the field along the prevailing wind erosion direction
V = equivalent vegetative cover

4.4.1 Soil

Soil erodibility refers to the ease of detachment and transport by wind. From wind tunnel tests, Chepil (1950) determined relative erodibility of soils reasonably free from organic residues as a function of apparent specific gravity and proportions of dry soil aggregates in various sizes. Clods larger than 0.84 mm in diameter were non-erodible in the tests. Since then, the non-erodible soil fraction greater than 0.84 mm, as determined by dry sieving, has been used to indicate erodibility of soil by wind (Chepil and Woodruff 1954). A dimensionless soil erodibility index, I, (Chepil and Woodruff 1954, 1959) was based on the non-erodible fraction (percentage of clods exceeding 0.84 mm diameter). The quantity of soil eroded in a tunnel is governed by the tunnel's length and other characteristics; therefore, erodibility was expressed on a dimensionless basis so that for a given soil and surface condition, the same relative erodibility value would be obtained regardless of wind tunnel characteristics (Chepil 1960). The soil erodibility index was expressed as

$$I = \frac{X_2}{X_1}$$

where X_1 is the quantity eroded from soil containing 60 % of clods exceeding 0.84 mm, and X_2 is the quantity eroded under the same set of conditions from soil containing any other proportion of clods exceeding 0.84 mm. Soil erodibility index, I, gave a relative measure of erodibility, but actual soil loss by wind was not known. When a field is smooth, bare, wide, unsheltered, and noncrusted, its relative erodibility is equivalent to the soil erodibility index defined the above equation.

Soil characteristics that affect erodibility of soil due to wind are texture, structure, and water content. Texture is considered as the most dominant factor, and it is commonly agreed that particles smaller than 0.25 mm and larger than 0.08 mm are most easily eroded by wind. Soils with characteristics like fine sand particles having low organic matter contents, single grains to massive in arrangement, with friable and nonsticky consistence, etc. are more easily subjected to wind erosion. Dong et al. (2000) classified the soil in the arid and semiarid areas of China into seven primary types according to material composition in relation to wind erosion: Gobi desert, sandy desert, loess deposits, residuum, flood deposits, salinized deposits, and irrigation deposits, and hypothesized that the sandy desert, desert/loess transitional zone, and flood deposits along abandoned river channels are the most easily eroded by wind. Chen (1991) suggested that soil structure, degree of compaction, and vegetation coverage are the main determining factors of soil resistance to wind erosion and found out that the threshold velocities of soil particles depend on the average diameter of soil aggregates instead of the grain size of single particles. After conducting experiments in a wind tunnel, Dong and Li (1998) discovered that the relationship for wind erodibility of aeolian sand as a function of its grain size follows a discontinuous function, with 0.09 mm sand being the most susceptible to wind erosion. The erodibility of aeolian sand can be divided into 3 categories: difficult to erode at >0.7 and <0.05 mm, moderately erodible at 0.7–0.4 mm and 0.075–0.05 mm, and most erodible at 0.4–0.075 mm. With similar grain size, a mixture of sizes is more susceptible to wind erosion than is a uniformly sized material (Skidmore 1974).

Chepil and Milne (1941) investigating the influence of surface roughness on intensity of drifting dune materials and cultivated soils found that the initial intensity of drifting was always much less over a ridged surface. Ridging cultivated soils reduced the severity of drifting, but ridging highly erosive dune materials was less effective because the ridges disappeared rapidly. The rate of flow varied inversely with surface roughness. Because it was difficult to determine surface roughness by measuring surface obstructions, Zingg and Woodruff (1951) devised a method to determine surface roughness from pressure relationships in a wind tunnel duct. The roughness was controlled by constructing ridges of non-erodible gravel. Armbrust et al. (1964) studied the effects of ridge roughness equivalent on total quantity of eroded material from simulated, cultivated soils exposed to different friction velocities. A ridge roughness equivalent of 6 cm reduces wind erosion 50 %. As roughness increases to about 11 cm, the soil ridge roughness factor remains about constant; then, with additional roughness, the effectiveness of ridges gradually decreases. As roughness increases to about 11 cm, the soil ridge roughness factor (K) remains about constant; then, with additional roughness, the effectiveness of ridges gradually decreases (Skidmore 1974).

4.4.2 Climate

Climate influences wind erosion by determining wind velocity, precipitation, and temperature that, in turn, determine evaporation and soil moisture content. Wind erosion occurs when the shear stress exerted on the surface by the wind exceeds the ability of the surface soil materials to resist detachment and transport. Strong winds erode, and dryness increases the susceptibility of the surface soil to erosion. The aridity of an environment is often evaluated by the Budyko dryness ratio (Budyko 1958; Hare 1983). The dryness ratio at a given site indicates the number of times the net radiative energy could evaporate the mean annual precipitation. Semiarid zones where wind erosion is likely to be a serious problem have a dryness ratio between 2 and 7 (Hare 1983). Areas with dryness ratios larger than 7 are in the desert and desert margin zones. Most of the Great Plains of the USA has dryness ratios between 2 and 5. The Sahara Desert in North Africa has a maximum dryness ratio as high as 200 (Henning and Flohn 1977). The higher the dryness ratio, the greater is the risk of wind erosion.

Chepil et al. (1962) considered the climatic elements wind velocity, precipitation, and temperature as the dominant factors relative to wind erosion rates and introduced a combined wind erosion climatic factor C, which was modified by Woodruff and Siddoway (1965) as

$$C = 34.483 \frac{v^3}{(P - E)^2}$$

where v means annual wind velocity for a particular geographic location corrected to a standard height of 30 ft, and $P - E$ = Thornthwaite's $P - E$ ratio = 10 $P - E$ = 115 $(P/T - 10)^{1.111}$. Later FAO (1979) revised the equation as

$$C = \frac{1}{100} \sum u^3 \left(\frac{ETP - P}{ETP} \right) d$$

where C is the wind erosion climatic factor, u (m s^{-1}) is mean monthly wind speed at 2 m height, ETP (mm) is potential monthly evaporation amount, P (mm) is monthly precipitation, and d is the number of days in the month concerned. Dong and Kang (1994) selected meteorological data from 233 stations covering parts of 12 provinces of northern China to calculate the wind erosion climatic factor in arid and semiarid China. The annual C value in arid and semiarid areas in China ranged from 10 to 150.

4.4.3 Field Width and Slope

It was shown by wind tunnel simulation (Dong 1994; Li 1999) that there are two effects of slope on wind erosion. Firstly, the wind velocity increases with slope along the upslope direction of a landform and strengthens the wind erosion on the slope, and secondly, the threshold velocity of grain increases with slope and weakens the wind erosion.

The rate of soil flow increases with distance downwind across an eroding field until, if the field was large enough, it reached a maximum that a wind of a given velocity can carry. Beyond that point the rate of flow remained essentially constant (Chepil 1957a). This maximum was about the same for soil of any texture about 2 t per rod width per hour for a 40 mph wind at 50 ft. The distance required for soil flow to reach the maximum that a wind of a given velocity can carry varies inversely with erodibility of a field surface. The more erodible the surface, the shorter the distance to reach maximum flow (Chepil and Woodruff 1959).

4.4.4 Vegetation

Vegetative covers reduce the wind velocity at the soil surface and also generally decrease the soil erodibility. The relationship between vegetation coverage and wind erosion rate is an exponential function, i.e., with the increase of vegetation coverage the wind erosion rate decreases exponentially. The measurements of threshold velocity and wind erosion in wind tunnel tests under various vegetation conditions showed that the threshold velocity increases with vegetation coverage, and that wind erosion rate decreases sharply as vegetation coverage increases (Liu et al. 1992). Quantitative relationships between the crop residues and wind erosion were reported early by Chepil (1944). Several latter workers observed that crop residues and stubbles effectively reduce wind erosion. Amounts of wheat straw needed to protect most erodible dune sand and less erodible soils against strong winds were established. Standing stubble is much more effective than flattened stubble. Standing sorghum stubble with rows perpendicular to wind direction controlled wind erosion much more effectively than rows parallel to wind direction (Englehorn et al. 1952). Siddoway et al. (1965) quantified the specific properties of vegetative covers influencing soil erodibility and developed regression equations relating soil loss by wind to selected amounts, kinds, and orientation of vegetative covers, wind velocity, and soil cloddiness. They found a complex relation between the relative effectiveness of different kinds and orientation of residue. The relative value of kinds and orientations of residue to control erosion must be qualified by soil, wind velocity, and variable characteristics of the residues. Generally they concluded that (1) fine-textured residues are more effective than coarse-textured residues, (2) any orientation of residue except flattened decreases wind erosion, and (3) fine-leafed crops, like grasses and cereals, provide a high degree of erosion control per unit weight. Craig and Turelle (1964) presented equivalent vegetative cover for additional crops, including a figure for converting quantity of various crop residues (peanuts, soybeans, shredded cotton, sesame, standing cotton stalks) to quantity of equivalent flat small grain residue. Hayes (1966) suggests that if any residue is not represented, a curve for a residue most like it can be used.

4.5 Principles of Wind Erosion Control

As early as in 1910, a USDA Farmers' Bulletin listed actions to control soil blowing as follows:

- Increase the water content of the soil.
- Increase the amount of humus (organic matter).
- Provide a cover of growing vegetation.
- Leave the stubble of the previous crop still standing on the land.
- Provide an artificial cover of straw and brush lines.
- Plant windbreaks to protect fields.
- Leave the soil surface in small clods instead of in a finely pulverized condition.
- Roughen the surface by proper cultivation at right angles to the direction of dangerous winds.

Lyles et al. (1983) extracted the following four principles of wind erosion control from the above list:

1. Establish and maintain vegetation or vegetative residues.
2. Produce or bring to the surface non-erodible aggregates or clods.

3. Reduce field width along the prevailing wind direction.
4. Roughen the soil surface.

Again, Woodruff et al. (1972) mentioned the following principles: producing a rough, cloddy surface, reducing effective field width with barriers, and establishing and maintaining sufficient vegetative cover.

4.6 Wind Erosion Control Measures

Many conservation practices can be implemented to control wind erosion. Conservation practices are designed to either reduce the wind force at the soil surface or create a soil surface more resistant to wind forces. Some practices also trap saltating particles to reduce the abrasion of soil surfaces downwind.

4.6.1 Stabilization of Soil

Various soil stabilizers have been employed for the control of wind erosion (Chepil et al. 1963; Armbrust and Dickerson 1971; Lyles et al. 1974; Armbrust and Lyles 1975). Many of these products successfully control wind erosion for a short time. However, easy and inexpensive stabilization may also be obtained by vegetative measures such as wheat straw anchored with a rolling disk packer (Chepil et al. 1963). Often, chemical soil-stabilizing agents are used with varying success. The criteria for surface soil stabilizers according to Armbrust and Lyles (1975) are as follows: (1) 100 % of the soil must be covered, (2) the stabilizer must not adversely affect plant growth or emergence, (3) erosion must be prevented initially and reduced for the duration of the severe erosion hazard, usually for at least 2 months each season, (4) the stabilizer should apply easily and without special equipment, and (5) cost must be low enough for profitable use. They found five polymers and one resin-in-water emulsion that met all these requirements. These were Coherex, DCA-70, Petroset SB, Polyco 2460, Polyco 2605, and SBR Latex S-2 105. These stabilizers may prevent wind erosion if applied to the total soil surface and at a sufficiently high rate, but their costs are prohibitive. For applying soil stabilizers in agricultural land, we need to develop (i) methods for applying large volumes rapidly, (ii) reliable preemergent weed control chemicals for use on coarse-textured soils, (iii) films strong enough to withstand raindrop impact and still allow water and plant penetration, and (iv) films that at have no adverse effects on the soil–water–air environment.

4.6.2 Cover Crops

Cover crop means plants or a green manure crop grown for seasonal soil protection or soil improvement. Cover crops help control soil movement and protect the soil surface between crops. Cover crop reduces wind erosion by shielding the soil with

Fig. 4.3 Cover crop of clover in an orchard in California (Photo courtesy of USDA-NRCS)

vegetation and anchoring the soil with roots. Green manuring cover crops are tilled into the ground in the spring, at least 1 month before planting the next crop. This provides additional nutrients to the crop, as the cover crop decomposes. The Dust Bowl (a period of severe dust storms causing major ecological and agricultural damage to American and Canadian prairie lands in the 1930s) has taught farmers the importance of planting cover crops for the control of wind erosion. Legumes, such as soybean or clover, are common choices for cover crops. Their vegetation reduces ground air pressure, and their roots hold the soil in place, in addition to their contribution to fixation of nitrogen (Fig. 4.3).

4.6.3 Ridging and Surface Roughening

Chepil and Milne (1941) observed that the initial intensity of drifting was always much less over a ridged than a smooth surface. Ridging cultivated soils (Fig. 4.4) reduces the severity of drifting. However, ridging highly erosive dune materials was less effective because the ridges disappeared rapidly. Experimental data showed that the rate of flow varied inversely with surface roughness. It is influenced by ridge spacing and ridge height, and it is defined relative to a 1:4 ridge height to ridge spacing ratio. A soil ridge roughness of 6 cm reduces wind erosion 50 %. Fryrear (1984) found a greater reduction in wind erosion by ridging; erosion remained relatively constant as ridge roughness increased beyond 11 cm. Lyles and Tatarko (1982) found that chiseling of growing winter wheat on a silty clay soil increased greatly non-erodible surface aggregates without influencing grain yields. Listers, chisels, cultivators, one-way disks with two or three disks removed at intervals, and

Fig. 4.4 Ridging of soil for wind erosion control (Photo courtesy of USDA-NRCS)

pitting machines can be used to bring compact clods to the surface. Emergency tillage is most effective when done at right angles to the prevailing wind direction. Because clods eventually disintegrate (sometimes rapidly), emergency tillage offers, at best, only temporary wind erosion control (Woodruff et al. 1972).

4.6.4 Residue Management

Residues of the harvested crops protect soil against wind erosion. Standing crop residues provide non-erodible elements that absorb much of the shear stress in the boundary layer. When crop residues are sufficiently high and dense to prevent intervening soil surface drag from exceeding threshold drag, soil will not erode. Standing stubble effectively protects the soil from wind erosion. Stubble mulching is a crop residue management system using tillage, generally without soil inversion and usually with blades or V-shaped sweeps (McCalla and Army 1961; Mannering and Fenster 1983). The goal is to leave a desirable quantity of plant residue on the surface of the soil at all times. Residue is needed for a period of time even after the crop is planted to protect the soil from erosion and to improve infiltration. The residue used is generally that remaining from a previous crop. Figure 4.5 shows a ridged field retaining sufficient residues of the previous crop, and Fig. 4.6 shows that the residues of the past wheat crop have formed an effective cover mulch.

Any crop residue, either grown in place or hauled in and spread, can control wind erosion. Last year's wheat or corn residues are uniformly spread over the field; they

Fig. 4.5 Residue management in a ridged field (Photo courtesy of USDA-NRCS)

Fig. 4.6 Residues of the previous wheat crop form effective mulch (Photo courtesy of USDA-NRCS)

form effective cover mulch. However, residues must be spread and anchored to the soil surface by a packer or an anchoring agent such as cutback asphalt or asphalt emulsion. Depending on residue type, minimum amounts needed to control wind erosion are 5–10 t ha^{-1}.

Fig. 4.7 Perennial grass barriers for wind erosion control (Photo courtesy of USDA-NRCS)

4.6.5 Wind Barrier

Barriers reduces wind erosion by (1) reducing the field width, (2) reducing the distance that wind travels in crossing unprotected field strips, (3) decreasing wind velocity, and (4) trapping wind-blown and saltated soil. Use of wind barriers is an effective old wind erosion control measure (Bates 1911). Different combinations of trees, shrubs, tall-growing crops, and grasses can reduce wind erosion. Besides the more conventional tree windbreak (Read 1964; Ferber 1969; Woodruff et al. 1976), many other barrier systems are used to control wind erosion. They include annual crops like small grains, corn, sorghum, Sudan grass, sunflowers (Carreker 1966; Fryrear 1969; Hagen et al. 1972), and tall wheatgrass (Black and Siddoway 1971; Aase et al. 1976) (Fig. 4.7). Most barrier systems for controlling wind erosion, however, occupy space that could otherwise be used to produce crops.

Typically windbreaks are linear plantings of single or multiple rows of trees or shrubs in boundaries of crop fields and farm holdings (Fig. 4.8). They are arranged usually perpendicular to the direction of the prevailing wind. They reduce the velocity of the wind at the ground level to a distance approximately ten times of the height of plants. Winter wheat, barley, rye, alfalfa, vegetables, and fruits can be successfully grown when fields are sheltered with windbreak. Sudmeyer et al. (2007) suggested that windbreaks of trees, or tree and shrub combinations, can offer many benefits on wheat belt farms, in addition to protection of soil, stock, crops, and pastures. Perennial barriers grow slowly and are often established with difficulty (Dickerson et al. 1976; Woodruff et al. 1976). A list of shrubs and trees suitable for windbreaks is given below.

Shrubs: American cranberry bush (*Viburnum trilobum*), dogwoods (*Cornus sericea*), (*Cornus racemosa*), (*Cornus alternifolia*), (*Cornus amomum*), silky elderberry

Fig. 4.8 Windbreak (Photo courtesy of USDA-NRCS)

(*Sambucus canadensis*), American hazelnut (*Corylus americana*), willow sandbar (*Salix interior*), silver buffaloberry (*Shepherdia argentea*), etc.

Small trees: American plum (*Prunus americana*), black chokeberry (*Aronia melanocarpa*), chokecherry (*Prunus virginiana*), crabapples (*Malus* sp.), Hawthorn, Arnold (*Crataegus arnoldiana*), Nanking cherry (*Prunus tomentosa*), pin cherry (*Prunus pensylvanica*), etc.

Deciduous trees: River birch (*Betula papyrifera*), autumn splendor (*Catalpa speciosa*), cottonwood (*Populus deltoides*), American elm (*Ulmus americana*), silver maple (*Acer saccharinum*), red oak (*Quercus rubra*), white oak (*Quercus alba*), black walnut (*Juglans nigra*), etc.

Conifers: Eastern red cedar (*Juniperus virginiana*), eastern white pine (*Pinus strobus*), ponderosa pine (*Pinus ponderosa*), red pine (*Pinus resinosa*), white cedar (*Thuja occidentalis*), white spruce (*Picea glauca*), etc.

4.6.6 Strip Cropping

The practice of farming land in narrow strips on which the crops alternate with fallow is an effective aid in controlling wind erosion (Chepil 1957a). Strips are most effective when they are at right angles to the prevailing wind erosion direction but also provide some protection from winds that are not perpendicular to the field strip. Strip cropping reduces erosion damage in the following ways: it reduces the distance the wind travels across exposed soil, localizes drifting that starts at a focal point, and reduces wind velocity across the fallow strip when adjacent fields are covered with tall stubble or crops.

4.7 Stabilization of Dunes

A dune is usually a low hill of sand built by the flow of wind or water. Dunes occur in different forms and sizes throughout the world, from coastal and lakeshore plains to arid desert regions. In addition to the remarkable structure and patterns, they are habitats of a variety of life adapted to this unique environment. Most kinds of dunes are longer on the windward side where the sand is pushed up the dune and have a shorter slip face in the lee of the wind. Dunes are sensitive and unstable ecosystems. Dunes need to be stabilized, and for it, every effort should be made to protect the integrity of the natural dune ecology.

Dune stabilization involves structural and vegetative measures. Vegetative measures are more effective to stabilize and rehabilitate the dunes. Vegetation establishment can be done by planting native grasses, trees, shrubs, or ground covers. The most commonly used dune grass in the USA is the American beach grass which has been effective in the initial stabilization of moving sand. Dune grass should be planted before the ground becomes frozen (in cold areas). Trees for dune stabilization include red pine, white pine, and jack pine. Trees and shrubs that can be used on wet spots of shifting sands are cottonwood, shrub willow, and dogwoods. Tree species adapted to sand dune plantings adjacent to large lakes are cottonwood, birch, white pine, white oak, and bigtooth aspen. In Mauritania several tree and grass species are used with success. *Prosopis juliflora* has been successful and sustainable, and *Aristida pungens* is planted on very mobile strip dunes. *Leptadenia pyrotechnica*, *Aristida pungens*, and *Panicum turgidum* are planted in deflation zones. Other woody species, mainly various acacias (especially *Acacia raddiana* and *A. senegal*), *Balanites aegyptiaca*, *Euphorbia balsamifera*, and *Persica salvadora*, are planted in more stable zones. Only halophytic woody and grassy species (those resistant to soil salinity and salt spray) are able to grow on coastal dunes. These include *Nitraria retusa*, *Tamarix aphylla*, *T. senegalensis*, *Casuarina equisetifolia*, *Atriplex halimus*, *A. nummularia*, and *Zygophyllum* spp.

Dune grasses can be effectively used for dune stabilization. Grass should be planted in a staggered or diamond pattern for maximum erosion control. Holes should be spaced 50 cm apart in areas where wind velocities and sand movement are high. Culm-to-culm distance should be 60 cm in areas not directly exposed to strong wind. The holes for plants should be between 15 and 25 cm deep to prevent the base of the stem from drying out and to prevent the entire plant from blowing out. Sand should be firm and moist around roots, with no air pockets near the base of the plants. Mulch should be applied between plants to protect plantings against rain and wind. Trees can be planted in beach grass after it has controlled sand movement, but before the grass becomes too dense. This may be done about 2 years after planting beach grass. Spacing of trees should be 2×2 m.

The mulch technique which consists of covering the dune uniformly with a natural or artificial protective cover to prevent saltation can be adopted on flat or reasonably even surfaces. The mulch can be made of various materials, such as straw, branches, stalks, plastic film or acrylic fiber, and mesh. Numerous studies have demonstrated that the straw checkerboards are efficient measures of dune stabilization. Figure 4.9

Fig. 4.9 Making straw checkerboard on dunes to stabilize them (Photo courtesy of Michael Reynolds)

shows the efforts of stabilizing sand dunes with wheat straw checkerboard to be followed by vegetation in Ningxia, China. The Autonomous Region Ningxia people have been combating against deserts and rolled them back by dwindling sand areas to 1.18 M ha from 1.65 M ha since the 1970s. At present, 470,000 ha of desert has been tamed by fixing sand with nets of wheat straw and planting drought-resistant plants. Ningxia is located in an arid and semiarid area. Straw mulching enhances cryptogam colonization and increases microorganism numbers, which are related to the formation of cryptogam crusts (Li et al. 2000). It increases the survival rate of shrub seedlings due to reduced soil erosion (Li et al. 2006) and as litter provides nutrients by decomposition (Hu and Zhou 1991). To date, the approach of using straw checkerboards and revegetation has been successful and effective for controlling mobile dunes and rehabilitating desert ecosystems (Fullen and Mitchell 1994; Whitford 2002; Li et al. 2004). Straw checkerboard and vegetation can be integrated. Figure 4.10 shows stabilized sand dunes in southern Saskatchewan, Canada.

Study Questions

1. Wind erosion is a problem of the arid and semiarid regions – why? What are the causes of accelerated wind erosion? What are the on-site and off-site effects of wind erosion?
2. Describe the processes of wind erosion. Mention the principles of wind erosion control.
3. Narrate the cover crops, grass barriers, and windbreak as wind erosion control measures.
4. Discuss residue management, stabilization of soil, and mulching in relation to wind erosion control.
5. What are dunes? Why should they be stabilized? Discuss the vegetative measures for dune stabilization.

Fig. 4.10 Stabilized dunes in Great Sand Hills, Saskatchewan, Canada (Photo courtesy of Visualphotos)

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Chapter 5
Chemical Soil Degradation

Estimates in 1991 suggest that about 240 M ha land is chemically degraded. Nutrient depletion has affected 136 M ha, salinization damaged 77 M ha, and acidification degraded 6 M ha. Another 11 M ha is affected by soil pollution. Agricultural mismanagement (58 %) and deforestation (28 %) are the main causes of chemical degradation of soil. Nutrient depletion is the most prevalent in Africa (65 M ha) and South America (68 M ha), while salinization is the major chemical degradation in Asia (53 M ha). Nutrient depletion has caused serious nutrient imbalances in soils under low-input agriculture in marginal lands. In a study, all the African agricultural soils exhibited negative NPK balances. Nutrient depletion is caused by leaching, residue harvest and burning, erosion, and crop removal. Salinization occurs naturally by pedogenic processes in different climatic conditions, but human-induced salinization has compelled to abandon many soils, which were productive earlier. The principal cause of human-induced salinization is inappropriate irrigation system in arid and semiarid regions. Leaching of salts by extra irrigation and growing salt-tolerant crops are the strategies for salty soil management. Soils are acidified by acid rains, base leaching, and by the use of acidifying fertilizers. Liming is an ancient method of reclaiming acidic soils. Growing crops suitable for the current soil pH may be profitable in low to medium acidic soils.

5.1 Chemical Degradation of Soil and Its Extent

Chemical degradation of soil refers to the undesirable changes in soil chemical behavior so that the quality of soil declines due to human interventions. Almost 240 M ha of land is affected by chemical degradation including 229 M ha in loss of nutrients (136 M ha), salinization (77 M ha), and acidification (6 M ha). Another 11 M ha is affected by soil pollution which is described in Chap. 6. Chemical soil pollution accounts for about 12 % of the total area affected by human-induced soil degradation (Oldeman 1994). In Africa, Asia, Central America, and Europe, about 12 % of the degraded soils are chemically affected. In South America, about 30 %

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Table 5.1 Global extent of soil degradation (M ha) by loss of nutrients, salinization, and acidification

Regions	Loss of nutrients	Salinization	Acidification	Total
Africa	65	15	1	62
Asia	15	53	4	72
South America	68	2	–	70
Central America	4	2	–	6
North America	–	+	+	+
Europe	3	4	+	26
Oceania	+	1	–	1
World	136	77	6	229

Data extracted from Oldeman (1994)

of degraded soils are due to chemical degradation. In North America and Oceania, chemical degradation of soil is of little concern. Loss of nutrients is the major type of chemical soil degradation in Africa and South America, while salinization is the major type of soil chemical degradation in Asia. Chemical soil degradation is mainly caused by agricultural mismanagement (56 %) and deforestation (28 %) (Oldeman 1994). Table 5.1 shows the global extent of chemical degradation of soil.

5.2 Loss of Nutrients and/or Organic Matter

Although it is said that the loss of nutrients is a problem with low-input agriculture, it also occurs in several other situations. It may occur if agriculture is practiced on poor or moderately fertile soils, without sufficient application of manure and fertilizers. The rapid loss of organic matter and nutrients of the topsoil occurs due to clearing of the natural vegetation. Burning of biomass, shifting cultivation, and erosion are other important causes of loss of nutrients and organic matter. Human-induced soil nutrient depletion is the process by which the soil nutrient stock is shrinking due to continuous nutrient mining without sufficient replenishment of nutrients harvested in agricultural products and of nutrient losses by accelerated soil erosion and leaching (Tan et al. 2005). Natural erosion and leaching are exacerbated by human actions for the exploitation of vegetation and soil resources. The general cause of fertility depletion is a negative balance between output (through harvesting, burning, leaching, etc.) and input (through manure and fertilizers, returned crop residues, flooding, etc.) of nutrients and organic matter. The nutrient balance situations in some African countries are cited as an example below. International Fertilizer Development Center (IFDC) conducted some studies of soil nutrient balances in some African countries. The methodological approach to estimate nutrient balances, depletion rates, and requirements combines information on agricultural production, soil characteristics, and biophysical constraints with methods and procedures designed for making such estimates (Henao and Baanante 1999). The information and data related to agricultural production included land use, population-supporting capacity of land, crop production, and use of mineral and organic fertilizers. The approach

Table 5.2 Nutrient balances in some African countries

Countries	Nitrogen (kg ha^{-1} $year^{-1}$)	Phosphorus (kg ha^{-1} $year^{-1}$)	Potassium (kg ha^{-1} $year^{-1}$)
Benin	−16	−2	−11
Botswana	−2	0	−2
Cameroon	−21	−2	−13
Ethiopia	−47	−7	−32
Ghana	−35	−4	−20
Kenya	−46	−1	−36
Malawi	−67	−10	−48
Mali	−11	−2	−10
Nigeria	−37	−4	−31
Rwanda	−60	−11	−61
Senegal	−16	−2	−14
Tanzania	−32	−5	−21
Zimbabwe	−27	2	−26

Source: http://www.fao.org/docrep/006/y5066e/y5066e06.htm

uses attribute and geographic database systems in conjunction with empirical and mechanistic models to produce information for analyses and monitoring. The approach builds upon previous work on nutrient balances (Stoorvogel and Smaling 1990; Smaling and Fresco 1993; Smaling et al. 1993). The nutrient balance values are given in Table 5.2.

Soil nutrient depletion is linked to food insecurity in developing and least developed countries due to the intensification of land use for agricultural production without proper application of external inputs (Henao and Baanante 1999). The absence of nutrient replenishment of nutrient depleted soils is jeopardizing agricultural sustainability in these regions (Sheldrick et al. 2002). Long-term decline in crop yields under conditions of low-input and unbalanced fertilization in many parts of Africa, Asia, and Latin America (FAO/UNDP/UNEP/World Bank 1997). Stoorvogel et al. (1993) and Smaling et al. (1993) used nutrient budget approach to highlight the serious situation of soil nutrient depletion in many African countries. Soil nutrient depletion and imbalance are also widespread problems in Asia. Dobermann et al. (1995) computed nutrient balances for rice-based farming systems in several eastern Asian countries and reported that the K deficit ranged from 25 to 70 kg ha^{-1} $year^{-1}$. Lin et al. (1996) estimated nutrient balances on 71 rice paddy farms in South China and found a surplus of N and P but a deficit of K. Mutert (1996) reported negative balances of N, P, and K in Bangladesh, Indonesia, Myanmar, the Philippines, Thailand, and Vietnam.

5.2.1 Leaching of Nutrients

Leaching is the downward movement of soil materials in solution and suspension with percolating water. It distributes and removes significant quantities of materials provided that there is a positive water balance, that is, if there is enough water excess of evapotranspiration for the effective downward movement. So, leaching is an important process of nutrient removal in the humid regions. Leaching occurs even in natural vegetation, but agricultural activities can greatly increase leaching losses (Havlin et al. 1999). Soils with high water infiltration rates and low nutrient retention capacity such as sandy soils and well-structured ferralitic soils (Oxisols, Ultisols) with low activity clays and low organic matter contents are particularly susceptible to nutrient leaching. Nutrients differ in their mobility in soil; nitrates are more mobile than other anions. Nitrate ions are not attracted by the negatively charged matrix of the top soil. Nitrates are continuously produced in soil by nitrification of ammonia obtained from fertilizers and soil organic matter. Nitrate leaching is a serious problem in sandy and low retentive soils (Robertson 1989). It often pollutes the groundwater and water of open reservoirs around agricultural landscapes. Sulfates are also leached easily from agricultural soils, but phosphates are relatively immobile. Soil colloids and calcium, aluminum, iron, and manganese immobilizes phosphates. The soil solution that percolates through the soil profile is essentially electrically neutral; it contains anions and equivalent amounts of cations. So, cations are also leached, and the more susceptible cations are calcium and magnesium than potassium (Pieri 1989). In sandy soils, considerable amounts of magnesium can be leached after applications of potassium chloride or potassium sulfate fertilizers (Havlin et al. 1999). Potassium is leached in much lower amount than calcium and magnesium.

5.2.2 Nutrient Loss due to Residue Burning

When biomass residues, crop residues or litter, are burned, most nitrogen (N) and sulfur (S) in the residue are lost, while mineral nutrients, such as phosphorus (P) and potassium (K), are retained. A Manitoba laboratory studied wheat, oat, and flax residue burned in an uncovered container (Heard et al. 2001). After the burn, the remaining ash was collected and weighed. Much of the N and S was oxidized and lost as volatile gases, while the mineral elements (P and K) remained in the ash. The study showed N losses of 98–100 %, S losses of 75 %, P losses of 21 %, and K losses of 35 %. It was concluded that the P and K loss resulted from smoke and ash that escaped from the burn container. In a field burn, some of this ash may be redeposited onto the field, depending on wind and other environmental factors. Other mineral nutrient loss could be assumed to be similar to P and K. The weather conditions after the fire often play an important role in nutrient loss from the field. For example, high winds can blow ash from the field or can pile the ash into drifts. Shifting cultivators burn their biomass residues to supply nutrients to their crops. Such nutrients have actually temporary benefit because heavy rains following land preparation often remove the bases released due to burning. In addition, soil erosion from both wind and water can occur more readily when the residue cover is destroyed and the soil is left exposed. A western Canada study where the cereal residue was burned annually for 19 years showed average annual soil organic matter declines of 0.03 and 0.07 % (Biederbeck et al. 1980).

5.2.3 *Nutrient Losses due to Erosion*

Amounts of nutrient losses by water erosion are affected by soil type, cover, and fertility; slope of the terrain; amount, method, and time of fertilizers application; intensity, amount, and moment of occurrence of rainfall (Gascho et al. 1998); as well as by soil management practices (Barisas et al. 1978; Schick et al. 2000; Mello 2002). Nutrient losses are expected to be reduced in soil management systems that preserve plant residues. However, even under such conditions, plant residues can be washed off (Burwell et al. 1975), becoming sources of soluble nutrients (Barisas et al. 1978; Johnson et al. 1979), which can be lost by water erosion. Ca and Mg losses by water erosion are usually high (Bertol 1994; Schick et al. 2000). This occurs because these elements are strongly adsorbed by colloids in the soil, which makes it easy for them to be transported with the sediment, and also because they occur in high concentrations in erosion water. Gafur et al. (2000) reported loss of 61 kg Ca, 13 kg Mg, 13 kg K, 0.14 kg P, 0.20 kg S, 0.05 kg Cu, 6.7 kg Fe, 6.1 kg Mn, and 0.065 kg Zn due to erosion in slashed and burnt sites in southeastern hills of Bangladesh.

5.2.4 *Crop Removal of Nutrients*

Crop removal of nutrients is affected by soil and climatic conditions. Soil properties such as available soil nutrient supply, moisture content, aeration, compaction, soil temperatures, pH, nutrient imbalances, and other factors affect uptake of plant nutrients. Table 5.4 gives the values of nutrients removed by the harvested crops. If residue contents of nutrients were added, these values could be multiplied by a factor of 2–3. So, crops alone remove a substantial amount of soil nutrient which need to be replenished by fertilizers. If residues are also removed from field, the loss becomes much higher. On an average, harvested crops remove 50–100 kg ha^{-1} N, 10–20 kg ha^{-1} P, and 15–30 kg ha^{-1} K in field crops. Potassium removal amounts to 100–200 kg ha^{-1} by forage crops (Table 5.3).

5.3 Acidification

Soil acidity is indicated by the concentration of hydrogen (H^+) ions in soil solution. When (H^+) ions predominate over (OH^-) ions, the soil is said to be acidic. The predominance of (H^+) and (OH^-) ions is expressed by soil pH. The pH is the negative logarithm of the hydrogen concentration (moles per liter), and its scale ranges from 1 to 14. A pH of 7.0 is taken as the neutral point, with values below 7.0 being acidic and above 7.0 being alkaline. Because the pH scale is logarithmic, soil with a pH of 5 is 10 times more acidic than soil with a pH of 6 and is 100 times more acidic than soil with a pH of 7. There are two types of soil acidity: active acidity created by the predominance of (H^+) ions in solution and reserve or exchangeable

Table 5.3 Nutrient removal by harvested crops (kg ha^{-1})

Crops	N	P	K	S
Grain crops				
Spring wheat	60–75	10–12	15–17	5–6
Winter wheat	60–65	11–13	14–17	7–9
Barley	80–95	15–18	20–25	7–9
Oats	60–76	12–14	15–18	5–6
Rye	60–75	11–14	15–20	5–6
Corn	95–145	18–22	22–28	7–8
Oilseed crops				
Canola	85–100	16–20	15–18	12–14
Flax	60–70	7–8	12–15	6–7
Sunflower	65–75	7–8	10–12	5–6
Pulses				
Pea	135–145	14–18	30–35	6–7
Lentil	62–74	8–10	27–35	4–5
Other crops				
Sugar beets	85–110	16–22	120–150	12–14
Potatoes	125–155	16–20	185–225	11–13
Forage crops				
Alfalfa	290–350	30–37	260–320	27–33
Clover	220–260	25–30	175–218	10–12
Barley silage	145–220	22–30	110–130	14–21
Corn silage	170–200	27–35	175–220	12–14

Converted from data compiled by the Canadian Fertilizer Institute from agronomic information obtained in Canada, 1998;
http://www.cfi.ca/_documents/uploads/elibrary/d161_NU_W_01%5B1%5D.pdf. Accessed 3.1.2012

Table 5.4 Aluminum sensitivity/tolerance of some plants

Categories	Plants
Highly sensitive	Durham wheat, barley, lentils, chickpeas, alfalfa, strawberry, berseem, buffel grass, tall wheatgrass
Sensitive	Canola, red clover, balansa clover, white clover
Tolerant	Whistler, diamond bird wheat, ryegrass, tall fescue, subterranean clover, chicory
Highly tolerant	Narrow leaf lupins, oats, triticale, cereal rye, cocksfoot, paspalum, yellow and slender serradella, Consol love grass

acidity characterized by the predominance of exchangeable H^+ and Al^{3+} ions on exchange sites of soil colloids. The larger the percentage of exchange sites occupied by aluminum and hydrogen, the lower is the pH and the higher is the acidity of the soil.

5.3.1 Causes of Soil Acidification

Acidification can occur naturally in soils developed from acidic parent materials due to release of acid-forming chemical compounds, in high rainfall areas due to prolonged leaching of bases, at higher elevations due to erosion, and under intensive weathering over a long period of time due to predominance of iron and aluminum oxides. But agricultural practices may accelerate or create soil acidity within a short time. Acidification in agricultural soils may be due to application of nitrogenous fertilizers, leaching of nitrates, removal of produce, and buildup of soil organic matter (Upjohn et al. 2005).

5.3.1.1 Fertilizers

The amount of acidification that results from using nitrogenous fertilizers depends on the fertilizer type (Table 5.4). Fertilizers that contain nitrogen as ammonium, for example, ammonium sulfate, acidify the soil within weeks after application. The most important acid-forming reaction for fertilizers is microbial oxidation of ammonical fertilizers, which may themselves be strong bases, by the following reactions:

1. $NH_3 + 2O_2 = H^+ + NO_3^- + H_2O$ (nitrification of ammonia)
2. $NH_4NO_3 + 2O_2 = 2H^+ + 2NO_3^- + H_2O$ (nitrification of ammonium nitrate)
3. $CO(NH_2)_2 + 4O_2 = 2H^+ + 2NO_3^- + H_2O + CO_2$ (hydrolysis of urea and nitrification of products)

The materials above are the most common synthetic N inputs to agroecosystems, either as single or mixed NPK fertilizers, and will oxidize to the equivalent of nitric acid under the well-drained, aerobic conditions of most dryland agricultural soils. Calcium nitrate and sodium nitrate have a neutralizing effect on soil acidity, unless all the nitrate is leached, but they are expensive and use is restricted to horticulture. Using superphosphate fertilizer on crops and pastures does not directly acidify the soil. Applying pure sulfur or “flowers of sulfur” will acidify the soil.

A long-term experiment was laid out in 1962 by Dr. Lloyd A. Peterson at the Arlington Agricultural Research Station, located in South Central Wisconsin, as a 4×4×4 NPK fertility trial. After 30 years of fertilizer additions, soil samples were analyzed and reported by Barak et al. (1997). Application of ammonium nitrate and urea N fertilizers at the long-term fertility trial at Arlington, Wisconsin, had caused soil acidification. Mean treatment values of pH ranged from 5.6 to 4.8 when measured in water and from 5.5 to 4.1 when measured in 1 M KCl for 0–150 lb N/acre annually treatments, respectively. A list of fertilizers with varying acidifying potential is given below.

Slightly acidifying fertilizers: Urea, ammonium nitrate, and urea ammonium nitrate solutions

These products are slightly acidifying because they contain ammonium or produce ammonium when applied to the soil. For every kilogram actual nitrogen applied, 1.8 kg of pure calcium carbonate is required to neutralize the acidity.

Moderately acidifying fertilizers: Diammonium phosphate (DAP)

Diammonium phosphate is the primary phosphorous fertilizer available in many countries and has a moderate acidifying effect when applied. For every kg of N added as DAP, 5.25 kg of limestone is needed for complete neutralization. If phosphorus fertilizer is recommended, triple superphosphate (0-46-0) can be used; it does not impact soil pH.

Severely acidifying fertilizers: Ammonium sulfate, monoammonium phosphate (MAP)

These fertilizers are very acidifying and should be avoided if possible. Approximately 8 kg of agricultural limestone is needed to neutralize the effects of 1 kg of these fertilizers. For example, if the grower applied a 50 kg MAP or ammonium sulfate, 400 kg of lime would have to be applied. Usually farmers apply the fertilizers, but not the lime. So, the soil becomes acid in the long run. The amounts of lime required for unit amount of some fertilizers are shown below.

Nitrogen source	Lime requirement, kg pure $CaCO_3$ per kg N applied (Glendinning 2000)
Ammonium sulfate	5.2
Anhydrous ammonia	1.8
Ammonium nitrate	1.8
Urea	1.8
MAP	5.0
DAP	3.1

5.3.1.2 Leaching of Nitrate

Leaching of nitrate is a major cause of agricultural soil acidification. Nitrate is derived from the biological oxidation of ammonium or added as nitrate containing fertilizers. Ammonium is obtained in soil from fertilizers and mineralization of organic matter. The chemical processes that produce nitrate nitrogen from ammonium leave the soil slightly more acidic. This acidity is neutralized by plants discharging an alkaline substance as they take up nitrate nitrogen. While the plants continue to take up all the nitrate nitrogen, the acid/alkali balance of the soil surrounding the roots remains in balance. Leaching breaks the balance of the acid/alkali processes and results in increased soil acidity. Nitrate leaching has important environmental impacts. Leached nitrates from agricultural lands may pollute ground and surface water.

5.3.1.3 Removal of Produce

Grain, pasture, and animal products are slightly alkaline, and continued removal will lower the soil pH over time. If very little produce is removed, such as in wool production, then the system remains almost balanced. Where a large quantity of produce is removed as in the case of hay making (particularly clover or alfalfa hay), the soil is left significantly more acidic. Removal of produce by burning, for example, burning of stubble, does not change the acid/alkali balance of the soil, but gives a redistribution, leaving alkali at the soil surface as ash. If the ash then gets washed away, as might occur by rains following a fire, this would leave the soil more acidic.

5.3.1.4 Buildup of Soil Organic Matter

While increasing organic matter has many benefits, including improvement of soil structure, it also increases soil acidity. The acidification caused by a buildup in organic matter is not permanent and can be reversed if the organic matter breaks down. However, there will be a permanent change in the acid status of the soil if the topsoil containing the organic matter is eroded or removed.

5.3.1.5 Acid Rain

Watmough and Dillon (2003) reported that depletion of nutrient-base cations (Ca, Mg, K, and others) in soils occurs due to acid deposition. Acidification and depletion of nutrient cations due to acid deposition have been reported for several forested regions in North America. Continuing regional inputs of nitrogen and sulfur are of concern because of leaching of base cations, increased availability of soil Al, and the accumulation and ultimate transmission of acidity from forest soils to streams (Fenn et al. 2006). Acid deposition is known as acid rain. It occurs when emissions of acidic materials such as sulfur dioxide (SO_2), ammonia (NH_3), nitrogen oxides (NO_x), and acidic particles from the combustion of fossil fuels and other industrial processes undergo complex chemical reactions in the atmosphere and fall to the earth as wet deposition (rain, snow, cloud, fog) or dry deposition (dry particles, gas). The effect of acid rain on the soil depends greatly on the ability of soils to neutralize the acid. Calcareous soils are most effective in neutralization of acidity in acid rains and prevent lowering of soil pH. Anthropogenic soil acidification due to acid rain is connected to high soil solution concentrations of SO_4^{2-} and NO_3^- (Matzner and Davis 1996).

5.3.2 *Effects of Soil Acidity*

Acidic soils may have some or all of the following problems: reduction in the amount of nutrients being recycled by soil microorganisms (e.g., nitrogen supply may be reduced); phosphorus in the soil may become less available to plants; induced deficiencies of calcium, magnesium, and molybdenum; aluminum and manganese toxicity to plants and microorganisms; and uptake of heavy metals by plants.

5.3.2.1 Nutrient Availability

Soluble and exchangeable ions are the available nutrient forms in soil. Soil pH regulates the solubility of elements and compounds in soil and governs the availability of plant nutrients in soil. When soil pH decreases from 6.5, that is, as the soil becomes more acidic, the solubility of some elements including Fe, Al, Mn, Cu, and Zn increases and of some others, for example, Ca, Mg, and Mo decreases. On the other hand, when soil pH rises from 7.0, solubility of Ca, Mg, and Mo increases and that of Fe, Al, Mn, Cu, and Zn decreases. Aluminum, although highly predominant in soil, is not an essential element for plants. In strongly acidic soils (pH below 5), solubility of Al along with Fe and Mn increases to such an extent that they become toxic to many plants. Moreover, plants may suffer from Ca and Mg deficiency in acidic soils. At low pH (<5.5), phosphorus is precipitated with Al, Fe, and Mn as their polyphosphates. At high pH (>8.0), P is precipitated with Ca. At both soil acidity and alkalinity, P availability is reduced to deficiency levels. Availability of P is usually higher in the pH range of 6.5 and 7.0. Boron deficiency may also occur in strongly acidic and strongly alkaline soils. Availability of macronutrients (Ca, Mg, K, P, N, S) and Mo and B is restricted at low pH. On the other hand, availability of most micronutrients (Fe, Mn, Zn, Cu, Co) increases at low pH. The most satisfactory plant nutrient levels occur at a pH range of 5.5–6.5.

5.3.2.2 Toxicity of Iron, Aluminum, and Manganese

Nutrient toxicity can occur in acidic soils when the pH is 4.8 or lower (Slattery et al. 1999). The two most important toxicities in acidic soils are those of aluminum (Al) and manganese (Mn) (Slattery et al. 1999). In strongly acidic soils ($pH_W < 4.3$), aluminum and manganese become more available in the soil solution and are harmful to plant roots. Aluminum toxicity is the most common plant symptom on acidic soils and causes root stunting (Slattery et al. 2000). Reduced root growth impedes nutrient and water uptake and results in decreased production. Some plants are more tolerant than others to high levels of Al in the soil solution. Important productive plants such as alfalfa, phalaris, canola, and barley are difficult to establish and grow in acidic soils. Both low pH and toxic aluminum (Yokota and Ojima 1995) irreversibly affect the establishment of alfalfa. The growing of deep-rooted perennial pastures (such as alfalfa and phalaris) is seen as an answer to slowing the acidification process. Aluminum may block the uptake of Ca and may precipitate P in plant body. It may interfere with P metabolism including ATP and DNA functions. Plant roots become stunted with little branching; root tip and lateral roots become brown. Plants suffering from aluminum toxicity produce symptoms in leaves that resemble P deficiency.

Toxicity by oxidized Fe frequently occurs in soils of pH below 4.0. Manganese toxicity is likely in soils having 200–5,000 mg kg^{-1} Mn. Manganese concentrations in the range of 0.2–12 mM have been reported to produce severe growth limitations in solution culture studies of species such as cotton (*Gossypium hirsutum* L.) (Kennedy and Jones 1991), sweet potato (*Ipomoea batatas* L.) (Mortley 1993),

Table 5.5 Critical levels of manganese for some plants (Upjohn et al. 2005)

Manganese tolerance category	Plants	Critical leaf Mn level mg kg^{-1}
Highly sensitive	Alfalfa, pigeon pea, barrel and bar medics	200–400
Sensitive	White clover, strawberry clover, chickpea, canola	400–700
Tolerant	Subclover, cotton, cowpea, soybean, wheat, barley, triticale, oats	700–1,000
Highly tolerant	Rice, sugar cane, tobacco, sunflower, most pasture grasses, oats, triticale, tiga, currency, cereal rye	>1,000

sorghum (*Sorghum bicolor*) (Mgema and Clark 1995), and wheat (*Triticum aestivum* L.) (Taylor et al. 1991). Aluminum sensitivity of some plants is shown in Table 5.4. Critical levels of Mn in some plants are shown in Table 5.5.

5.3.2.3 Plant Growth

A slightly acidic soil (a pH around 6.5) is the optimum requirement of most terrestrial plants. At this pH soil microorganisms are most active, and plant nutrients are readily available. At extreme alkaline and acid conditions, this delicate balance is disturbed, and plant nutrients that were in adequate supply can become either deficient or toxic to plant growth. Some essential nutrients such as phosphorous, calcium, magnesium, and molybdenum become unavailable if the soil pH becomes too acid. Acid conditions will result in a lowering of plant production in farming systems leading to reduced profitability and an increased reliance on fertilizers to sustain any form of productive agriculture. Plants have pH preferences. A list of plants showing their suitable pH ranges is cited from Jett (2005) below:

pH range	Suitable crops
4.5–5.5	Azalea, Camellia, Rhododendron
5.5–6.0	Blackberry, chicory, gooseberry, magnolia, pineapple, potato, oats, rosemary
5.5–6.5	Apple, avocado, barley, cranberry, capsicum, melon, turnip, strawberry, wheat
6.0–6.5	Broad bean, bean, carrots, lemon, lentil, olives, parsley, pear, pumpkin, raspberry, soybean, squash, watermelon, white clover
6.0–7.5	Asparagus, broad bean, broccoli, cabbage, cauliflower, celery, cherry, chrysanthemum, cucumber, dahlia, French bean, garlic, grape, horse radish, lettuce, mulberry, mustard, onions, peach, pear, peas, peppers, plum, radish, spinach, sweet corn, tomato, turnips
6.5–7.5	Alfalfa, asparagus, avocado, barley, sugar beet, beetroot, grape vine, mushroom, spinach

5.3.2.4 Microorganisms

Soil acidity limits *Rhizobium* survival and persistence. Elevated levels of aluminum are toxic to the growth of Rhizobia affecting nodule initiation and the nitrogen fixation process (Slattery et al. 2001). Many Victorian soils are found to be in a pH range where clover nodulation is reduced. It has a significant impact on the productivity of pasture and subsequent crop systems that rely on the biological input of nitrogen. At very low soil pH and under conditions of high aluminum solubility, aluminum can have a metagenetic effect on Rhizobia. These findings suggest that the microbial biodiversity might alter over time with changing soil pH. In highly weathered acidic soil, bacterial growth is inhibited by nutrient toxicities and low nutrient availability. Acidity changes the soil microbial community and decreases root and rhizosphere effects. It decreases organic matter decomposition and nutrient cycling (Siqueira and Morera 1997). Acidity decreases the activity of nitrifying bacteria which are responsible for the breakdown of organic matter into ammonium and nitrate for subsequent plant uptake. Fungi generally have a pH optimum below that of bacteria.

5.3.2.5 Soil Fauna

Soil acidity influences the delicate balance between groups of living organisms in the soil. Generally, the soil fauna has a significantly reduced capacity to cope with large changes in soil pH. Most macrofauna including deep-burrowing species such as worms and termites tend to decrease in abundance in acidic soil conditions with most activity being confined to the litter layer where the pH is significantly higher and usually alkaline. Mele and Carter (1999) have shown that a reduced abundance of earthworms is associated with acidic soil conditions. Some worm species are able to tolerate acidic soil conditions and play an active role in litter decomposition, but these worms are also highly affected by the moisture conditions surrounding them. Deep-burrowing and soil-eating worm species are unable to tolerate low soil pH conditions (Lavelle and Spain 1995).

5.3.3 Management of Acidic Soils

Despite major difficulties for agricultural use, acidic soils can be very productive if lime and nutrients are applied at proper time and quantity. For management of acidic soils, the pH need to be adjusted to a desired level by liming, following adequate irrigation and drainage and selecting suitable crops. Liming is an efficient way of increasing soil pH and reducing toxicity of iron, aluminum, manganese, and molybdenum. Liming also stimulates biological activity in soils and increases cycling of nitrogen, phosphorus, and sulfur. It increases P availability to plants. It improves soil structure, porosity, aeration, and water movement in soils. However, over liming reduces the availability of phosphorus and causes deficiencies of micronutrients such as manganese, zinc, copper, and molybdenum.

5.3.3.1 Liming Materials

Liming materials or lime are any materials that contain Ca or Mg and are able to neutralize soil acidity. Liming materials are generally carbonates and bicarbonates, sometimes oxides and hydroxides of calcium and magnesium.

Carbonates: The most widely used liming materials are the carbonates. They are less expensive and easier to handle than other lime materials. Ground high-grade limestone or calcite is concentrated calcium carbonate ($CaCO_3$). Dolomite ($MgCO_3 + CaCO_3$) is also a common source of agricultural lime. Dolomite usually costs a little more than calcite and changes the soil pH more slowly, but it has the advantage of containing Mg in addition to Ca. Together, calcite and dolomite account for more than 90 % of the lime used in agricultural fields. Both materials are naturally occurring rocks that are mined and ground for agricultural use. Marl and oyster shells are also carbonate materials. Marl is a naturally occurring mixture of clays, carbonates of Ca and Mg, and shell remnants. Oyster shells also contain high proportion of calcium carbonate but are important only in some coastal regions.

Oxides: Oxide liming materials include burned lime, unslaked lime, and quicklime. Oxides are made by baking crushed calcite limestone or dolomite limestone in a furnace, thereby driving off carbon dioxide (CO_2) to form a concentrated oxide (CaO or MgO). This material is of low molecular weight and reacts rapidly in the soil to raise the pH. Oxides are the most efficient of all liming materials on a pound-for-pound basis. But oxides are caustic; that is, they react with moisture and are difficult to handle. In addition, their cost is high relative to carbonate materials. One ton of calcium oxide has the neutralizing power of 1.8 tons of calcite.

Hydroxides: Hydroxides are simply oxide materials with water added. They are also known as hydrated lime, slacked lime, or builders lime. These materials are similar to oxides because they are powdery, quick acting, and unpleasant to handle. Hydroxides are also more expensive than carbonate materials.

Fluid Lime: Lime can be applied in liquid form as a suspension. Lime is not dissolved but is suspended in water. Liquid lime may be prepared by pouring 50 % water to 48 % lime solids and 2 % clay to maintain a suspension. The material used in this suspension should pass a 100-mesh sieve. The advantages of using liquid lime include (1) good application uniformity, (2) high-quality lime material (mesh size), and (3) quick soil pH change.

Miscellaneous Lime: Many byproducts of mining, refining, processing, and manufacturing processes are used as liming materials. Slags from blast furnaces and electric furnaces as well as fly ash and bottom ash from coal-burning plants are often applied as lime. Lime sludges from sugar beet processing plants (sugar lime), paper mills, ore processing, and water-softening plants are sometimes used to raise soil pH. Wood ashes from wood stoves or fireplaces also may be used. Active gradients in these materials are quite variable and their requirement cannot be precisely assessed. Their contaminants also restrict their use.

Table 5.6 Calcium carbonate equivalence of common liming materials

Liming material	Composition	CCE
Calcitic limestone	$CaCO_3$	100
Dolomitic limestone	$CaMg(CO_3)_2$	109
Calcium oxide	CaO	179
Calcium hydroxide or Magnesium hydroxide	$Ca(OH)_2$	136
Slag	$CaSiO_3$	80

Source: http://learningstore.uwex.edu/assets/pdfs/a3671.pdf

5.3.3.2 Quality of Lime

The quality of liming materials is determined by (1) the chemical composition or purity and (2) the particle size or fineness.

Chemical Composition

Acid-neutralizing capacity of a liming material is determined by its calcium carbonate equivalence (CCE). It is the acid-neutralizing capacity of the material compared to pure calcium carbonate. In CCE comparisons, pure calcium carbonate has been assigned a value of 100. CCE values of some common liming materials are given in Table 5.6.

Dolomite has a slightly greater CCE than calcium carbonate due to the lower atomic weight of Mg compared to Ca. Oxide materials have the highest CCE values since CO_2 is removed in the burning process. Marl and byproduct materials have low CCE values because of the presence of impurities.

Fineness

The neutralizing capacity of a liming material is determined by its particle size. 100 % of lime particles that pass a 100-mesh sieve will react within the first year while only 60 % of the liming materials that pass a 20-mesh sieve (but held on 100-mesh sieve) will react within a year of application. Materials that do not pass the 20-mesh sieve will not react within 1 year after application. So, for practical purpose, neutralizing power of a liming material needs adjustment for the fineness of the material. To determine the fineness factor of a liming material of which 70 % passes a 100-mesh sieve and 97 % passes a 20-mesh sieve, the following calculations need to be done:

(a) To subtract the percentage passing a 100-mesh sieve from the percentage passing a 20-mesh sieve and multiply this difference with 0.60
(b) To add the percentage passing the 100-mesh sieve and divide the sum by 100

Thus, the fineness of a material of which 70 % passes a 100-mesh sieve and 97 % passes a 20-mesh sieve is $\{(97-70)\times 0.60+70\}/100=0.86$.

Effective Neutralizing Value (ENV)

The ENV is the fraction of the liming material's CCE that will react with soil acidity in the first year of application. The ENV is calculated by multiplying the CCE with the fineness of the material. For example, a liming material with CCE of 90 % and a fineness of 0.86 has an ENV of 90 × 0.86 = 77.4. The neutralizing value of a liming material is its capacity to neutralize acidity. The higher the NV, the more pure is the product. Pure calcium carbonate (pure limestone) is taken as the standard with an NV of 100. The neutralizing value of commercial limestone is usually between 96 and 98. Other liming materials are more reactive than limestone and therefore have higher neutralizing values, for example, hydrated lime and burnt lime.

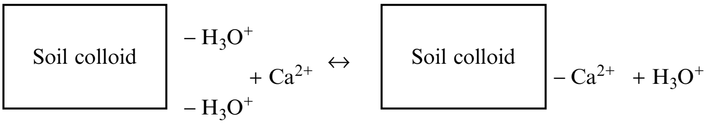
5.3.3.3 Mechanism of Lime Action

Lime dissolves slowly in soil solution to release Ca and bicarbonate ions:

$$CaCO_3 + H_2O + CO_2 = Ca^{2+} + 2HCO_3^-$$

The bicarbonate ions neutralize the hydronium ions in soil solution and the Ca^{2+} ions replace aluminum and hydronium ions from the exchange sites of the colloids. Once displaced into solution, these aluminum and hydronium ions are also neutralized by bicarbonate ions.

$$HCO_3^- + H_3O^+ = CO_2 + 2H_2O$$



5.3.3.4 Lime Requirement

Even if two different soils have the same level of acidity (say pH 5.5) and the cropping pattern needs to raise its pH to the same level, say 6.5, they will require different amounts of lime because the soils may differ in clay content, cation exchange capacity, and base saturation percentage. These properties give soils a unique capacity known as buffer capacity or the capacity to resist change in soil pH. Some of applied lime will neutralize acids in soil solution, and some will be consumed for replacing exchangeable H^+ and Al^{3+}. In other words, doses of lime cannot be estimated from the pH value alone. Lime requirement of each of the acidic soils needs to be determined separately. Nowadays computer programs are available to estimate lime requirement from existing pH, desired pH, clay content, CEC, exchangeable Al^{3+} and H^+, and percent base saturation.

There are a number of lime requirement determination methods for different types of soils. Two methods are generally used: the SMP (Shoemaker–McLean–Pratt) and Adams–Evans buffer methods. The SMP method was designed for use with soils that have large lime requirements and significant reserves of exchangeable Al (Shoemaker et al. 1961). The Adams–Evans buffer was designed for soils that are coarse textured, with low cation exchange capacities and organic matter contents, and thus low lime requirements (Adams and Evans 1962). The Mehlich lime buffer was developed for use on Ultisols, Histosols, Alfisols, and Inceptisols (Mehlich 1976). To avoid hazardous chemicals, all three buffers have been modified. All three modifications correlated well with the original buffer, and no changes in calibration were necessary for any of the modified methods (Hoskins 2005; Huluka 2005; Sikora 2006). The original SMP buffer or its modification (Sikora buffer) is the most commonly used method.

Lime Requirement Procedure (SMP Buffer Method)

SMP Buffer

1. Weigh 32.4 g of para-nitrophenol, 54.0 g of K_2CrO_4, and 955.8 g $CaCl_2 \cdot 2H_2O$ into an 18 L bottle. Add 9 L distilled water, shaking vigorously during addition.
2. Weigh 36.0 g of $Ca(OAc)_2$ into a large container and dissolve in 5 L of distilled water.
3. Combine the two solutions, shaking during mixing and every 15–20 min for 2–3 h.
4. Add 45 mL triethanolamine, shaking during addition and periodically thereafter until completely dissolved.
5. Dilute to 18 L with distilled water, adjust to pH 7.50 using 15 % NaOH, and filter.
6. Store in a container with the air inlet protected by drierite and ascarite to prevent contamination by water vapor and carbon dioxide. Avoid excessive agitation of the solution after pH adjustment.

Procedure

1. Add 10 mL of SMP buffer to the soil–water slurry used for pH determination.
2. Place in a mechanical shaker, close tightly, shake at 250 excursions/min for 10 min, and let it sit for 20 min.
3. Swirl, insert electrodes, and read the pH. Read to the nearest 0.01 pH unit.
4. Determine lime requirement from soil buffer pH and calibration data for local soils.

5.3.3.5 Applying Lime

Application of lime of proper quality and at appropriate rates on the surface of soil and incorporation is usually recommended. Good results have been achieved with deep placement of lime using modified farm-scale machinery as well, but it is difficult and time-consuming to achieve the desired distribution. Deep placement is only recommended for soils in which subsurface acidity is constraining production, and before it is attempted, detailed consideration should be given to whether it is likely to be profitable.

Surface application: The main aim of applying lime to the surface should be an even coverage of the ground. The spreading width should be approximately 6–8 m, depending on wind conditions to get good coverage of fine particles less than 0.5 mm. Spreading too wide results in uneven treatment of soil acidity. Larger particles will spread up to 15 m, but the effective distribution is poor and will result in variable change in pH.

Surface applied then deep-rip: Applying lime by spreading on surface prior to any soil disturbance and deep-ripping by plowing is a good practice. This enables better distribution of the lime particles and greater contact with the acidic soil. Liming before deep-ripping for compaction adds value to both treatments.

Shallow incorporation: Some farmers use shallow incorporation of 200–300 kg ha^{-1} of lime at seeding. Little work has been done to test whether this makes a difference to the rate or amount of amelioration that can be achieved. Research shows that low rates of lime that do not increase the surface pH to 5.5 or above are insufficient to prevent ongoing subsurface acidification.

Direct injection: Deep and direct injection involves specific machinery. Research has shown that successful direct injection is possible and subsurface acidity can be quickly removed as a production constraint. When the distribution of lime is correct, responses of 20–30 % in wheat are common. However, increased costs due to machinery modification and slow operation need to be considered. Poor distribution can result in the lime being placed below an untreated acidic layer, and root growth will still be restricted. Only where compaction is also a constraint may direct injection be worth considering.

5.3.4 Acid Sulfate Soils

Most acid sulfate soils occur in the tropics, in low-lying coastal lands formerly occupied by mangrove swamps. Their most important characteristics are a field pH of below 4, owing to the oxidation of pyrite to sulfuric acid, and generally high clay content. If samples of the pyrite layers are air dried in the laboratory, the pH may drop by a further 2 units. Potential acid sulfate soils have a near-neutral pH under field conditions but become strongly acid upon drainage and oxidation. The total area of actual and potential acid sulfate soils is rather small: about 10 M ha are known to occur in the tropics, and the world total probably does not exceed 14 M ha. In addition, some 20 M ha of coastal peats, mainly in Indonesia, are underlain by potential acid sulfate soil. Clearance of the native vegetation, deforestation, shrimp farming, etc. have converted many potential acid sulfate soils into extremely acidic soils. Potential acid sulfate soils can be used for satisfactory rice production, but when they are acidified, they are ultimately abandoned. The global distribution of acid sulfate soils as obtained from data of the FAO-UNESCO Soil Map is shown below.

Region	Area (M ha)
Africa	3.70
Asia and Far East	6.70
Latin America	2.10
North America	0.10
World total	12.60

5.4 Soil Salinization

A soil is said to be saline if the electrical conductivity of its saturation extract (EC_e) (saturating the soil with water and taking the extract by suction) is greater than 4 dS m^{-1} (decisiemens per meter). Saline soils contain excess soluble salts, generally chlorides and sulfates, with some carbonates and bicarbonates, of sodium, potassium, calcium, and magnesium. Soil salinity is harmful for plants barring the halophytes; it causes water stress through osmotic disturbances in plant tissue and by toxicity of some salt constituents. Seed germination, plant growth, and yield of crops are considerably reduced by soil salinity. Sometimes it causes crop failure.

Some soils are naturally saline. They are formed by processes called primary salinization or natural salinization. Some soils are made saline by mismanagement of soil and crop, particularly improper irrigation and drainage, that is, changing the hydrologic balance. This is known as secondary salinization or human-induced salinization. Salinization, both natural and human induced, may occur in two climatic settings – arid and semiarid and humid regions. In arid and semiarid regions, scarcity of water due to low rainfall and high evaporation does not allow necessary leaching of salts. Moreover, there is a net capillary rise of water which brings salts to the surface soil. In humid areas, on the other hand, excess irrigation or poor drainage cause the groundwater table to rise to the root zone of plants and make the soil saline.

Oldeman et al. (1991) estimated that worldwide 76.6 M ha lands are affected by human-induced salinization, but they did not differentiate salinity in the irrigated and non-irrigated rainfed areas. A survey made by Ghassemi et al. (1995) revealed that vast areas of India, China, Pakistan, and Central Asian countries are affected by salinity due to over irrigation and lack of drainage facilities. Some soils have become saline due to the use of saline water for irrigation. Table 5.7 gives areas affected by different degree of salinization in different continents.

Two important approaches to soil salinity management are (1) selection of salt-tolerant crops and (2) removal of excess salts from the root zone. Several salt-tolerant crops can be grown. Removal of salts by irrigation and drainage and drawing the salty water safely may be financially impracticable in some situations. Whether the reclamation will be cost-effective or not depends on salinity level, cost of water and labor, crop return, etc. Reclamation is relatively easy if the soil is saline alone, but difficult if it is saline – sodic or sodic.

Table 5.7 Global extent of human-induced salinization

Continent	Light (10^6 ha)	Moderate (10^6 ha)	Strong (10^6 ha)	Extreme (10^6 ha)	Total (10^6 ha)
Africa	4.7	7.7`	2.4	–	14.8
Asia	26.8	8.5	17.0	0.4	52.7
South America	1.8	0.3	–	–	2.1
North and Central America	0.3	1.5	0.5	–	2.3
Europe	1.0	2.3	0.5	–	3.8
Australia	–	0.5	–	0.4	0.9
Total	34.6	20.8	20.4	0.8	76.6

Oldeman et al. (1991), with permission

5.4.1 Management of Saline Soils

Saline soil management involves growing salt-tolerant crops and leaching of salts below the root zone. If there is a salt crust on the surface of the soil, salt farming may be more profitable than cropping. For cropping, decrusting may be done mechanically and with soil flushing. However, decrusting followed by flushing has not been very successful for increasing crop yields. But some soils may need decrusting before leaching. Effective leaching may be achieved by flooding and draining soils. Some soils are only slightly saline. For shallow-rooted crops, the salts may be driven below the root zone by temporary leaching. This technique will need less water than normal leaching. Furrow irrigation with suitable seedling placement may also be satisfactory. Drip irrigation at the root area also dilutes salts and keeps the salts apart. If the soil is considerably saline, it needs removal of excess soluble salts by thorough leaching. However, a reliable estimate of the quantity of water required to accomplish salt leaching is required. The salt content of the soil, salinity level to depth to which reclamation is desired, and soil characteristics are important factors determining the amount of water needed for reclamation. A useful rule of thumb is that a unit depth of water will remove nearly 80 % of salts from a unit soil depth. Thus, 30-cm water passing through the soil will remove approximately 80 % of the salts present in the upper 30 cm of soil. To leach soluble salts in irrigated soils, more water than required to meet the evapotranspiration needs of the crops must pass through the root zone to leach excessive soluble salts. This additional irrigation water has typically been expressed as the leaching requirement (LR). Leaching requirement was originally defined as the fraction of infiltrated water that must pass through the root zone to keep soil salinity from exceeding a level that would significantly reduce crop yield under steady-state conditions with associated good management and uniformity of leaching (Rhoades 1974):

$$LR = \frac{EC_{iw}}{EC_{dw}},$$

where LR is leaching requirement, $\mathrm{EC_{iw}}$ is electrical conductivity of irrigation water, and $\mathrm{EC_{dw}}$ is the electrical conductivity of drainage water. Several leaching

Fig. 5.1 Underground pipe drainage system removes leaching water (Photo courtesy of USDA-NRCS)

requirement models have been proposed by Corwin et al. (2007). Leaching is most often accomplished by ponding fresh water on the soil surface and allowing it to infiltrate. Effective leaching occurs when the salty drainage water is discharged through subsurface drains that carry the leached salts out of the area under reclamation. Leaching may reduce salinity levels when there is sufficient natural drainage, that is, the ponded water drains without raising the water table. Leaching should preferably be done when the soil moisture content is low and the groundwater table is deep. The ability to leach water through the soil profile is dependent upon good internal drainage. In fact, the overall effectiveness of a reclamation system is dependent upon leaching. Leaching carries the salts down through the soil profile and out of the rooting zone. Without it, salts will accumulate regardless of any applied soil amendments. If natural drainage is poor, then a tile or pipe drain (Fig. 5.1) should be installed under expert supervision of professionals. When properly installed, they are highly effective at carrying away the leached salts. Tile can only drain water if it is placed below the zone of saturation. Several methods of irrigation and drainage may be employed for salt and water management in saline soils. Drip and sprinkler irrigation methods are suitable in slightly saline soils.

5.5 Salt-Tolerant Crops

Some crops are tolerant to a level of soil salinity. It means that the yield of the crop is not reduced more than 25 % in the ECe level. Some crops are slightly tolerant; others are highly tolerant. Farmers should choose crops or crop sequences on the salinity values of their soils. Salinity tolerance of some field crops is given in Table 5.8.

Table 5.8 Salinity tolerance of some important crops

		Yield loss, ECe (dSm^{-1})		
Crop	Threshold value, ECe (dSm^{-1})	10 %	25 %	50 %
Asparagus	5.0	8.0	11.0	13.0
Barley	8.0	9.6	13.0	17.0
Beets	5.3	8.0	10.0	12.0
Beans	1.0	1.5	2.3	3.6
Broccoli	2.7	3.5	5.5	8.2
Cabbage	1.8	2.8	4.4	7.0
Canola	2.5	3.9	6.0	9.5
Carrot	1.0	1.7	2.8	4.6
Cauliflower	2.7	3.5	4.7	5.9
Celery	1.8	3.5	5.8	10.1
Corn	2.7	3.7	6.0	7.0
Cucumber	2.5	3.3	4.4	6.3
Lettuce	1.3	2.1	3.2	5.2
Oat	5.2	6.7	9.0	12.8
Onion	1.2	1.8	2.8	4.3
Peas	0.9	2.0	3.7	6.5
Potato	1.7	2.5	3.8	5.9
Rye	5.9	7.7	12.1	16.5
Safflower	5.3	8.0	11.0	14.0
Sorghum	4.0	5.1	7.1	10.0
Sugar beet	6.7	8.7	11.0	15.0
Sunflower	2.3	3.2	4.7	6.3
Triticale	6.1	8.1	12.0	14.2
Tomato	2.5	3.5	5.0	7.6
Wheat	4.7	6.0	8.0	10.0

Source: https://extension.usu.edu/files/publications/publication/AG-SO-03.pdf. Accessed on 15.8.2012

Study Questions

1. What do you mean by chemical degradation of soil? What are the different types of chemical degradation of soil? Discuss the state of chemical degradation of soil.
2. Explain nutrient depletion of soil. Why in many soils there are negative nutrient balances? Discuss nutrient leaching from soil.
3. Narrate natural and human-induced causes of soil acidity. Discuss Al and Mn toxicity on plants.
4. Explain liming, lime requirement, and quality of lime. How do you determine lime requirement of a soil? Discuss the factors affecting lime requirement of soil.
5. How does soil mismanagement create soil salinity? Why is soil salinity a prominent problem of arid and semiarid regions? Explain leaching requirement.

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Chapter 6
Soil Pollution

Soil pollution is the accumulation of a substance, native or introduced, in soil at a level harmful for the growth and health of organisms, including microorganisms, plants, and animals. Hazardous substances find their way to the soil with domestic, municipal, industrial, mining, and agricultural wastes and industrial and agrochemicals such as fertilizers and pesticides. The most important categories of soil pollutants are the persistent organic compounds such as PAHs, PCBs, PCNs, and PHEs and heavy metals such as Pb, Cd, As, Hg, Zn, and Cu. These substances, above a critical level, are toxic to plants and animals, including human. These soil pollutants can be removed by physical methods such as soil washing, encapsulation, and vitrification; chemical methods such as immobilization, precipitation, and oxidation; and biological methods such as microbial and phytoremediation. Hyperaccumulator plants are often employed for the remediation of soils polluted with heavy metals.

6.1 Pollution and Soil Pollution

Pollution is defined as the introduction of any substance to the environment that adversely affects the usefulness of a resource. The substance that is introduced and that adversely affects the environment is the pollutant. According to Van Der Perk (2006), pollution and contamination are two terms used synonymously to mean the introduction into the environment by humans of substances that are harmful or poisonous to people and ecosystems. According to many other authors, however, meanings of soil pollution and soil contamination are different. To them, soil contamination is the introduction to the soil of a chemical substance which was not originally present (de Haan and van Riemsdijk 1986); and soil pollution is the presence of a chemical substance in soil in concentrations higher than natural as a result of human activity and that has a detrimental effect on the soil environment and its components (Kabata-Pendias and Pendias 2001). For simplification, we use the term soil pollution to mean the presence of a substance, either introduced or native, above the threshold level. Soil pollution is a global problem that threatens the life of microorganisms,

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plants, animals, and humans. Soil pollution arises mainly from disposal of solid and liquid wastes in the ground; disposal of chemical wastes from the industries into the soil; indiscriminate use of agricultural chemicals, including fertilizers and pesticides; application of sewage sludge and compost without judging metal contamination; leakage of underground septic tanks into the soil; use of non-decomposable materials; mining activity; and irrigation with polluted water.

6.2 Sources of Soil Pollutants

According to Alloway (1995), various wastes, industrial, municipal, agricultural, domestic, and nuclear, are the chief sources of soil pollution. Wastes are materials that have reached the end of their useful life (Hill 2010). All wastes are not necessarily a pollutant. Many wastes can be recycled into useful materials again. Plastic and paper wastes can be converted to newer products. Excreta of animals are excellent manures. Crop residues can be effectively used for protection against erosion. However, most wastes are considered useless and often disposed of carelessly.

6.2.1 Municipal Waste

The DOE (2002) defines municipal waste as waste from households, as well as other waste, which, because of its nature or composition, is similar to waste from households. Municipal wastes include domestic and kitchen wastes, market wastes, hospital wastes, livestock and poultry wastes, slaughterhouse wastes, and metals, glass, and ceramic wastes. Municipal wastes can be grouped into five different categories: biodegradable (food and kitchen waste such as meat trimmings or vegetable peelings, yard or green waste, and paper), recyclable materials (glass, plastic bottles, other plastics, metals, and aluminum cans), inert waste (construction and demolition wastes; inert materials are those that are not necessarily toxic to all species but can be harmful or toxic to humans. Therefore, construction and demolition waste is often categorized as inert waste), composite waste (clothing and plastics), and hazardous wastes (medicines, paint, batteries, light bulbs, fertilizer and pesticide containers, and e-waste like old computers, printers, and cellular phones). Some of these wastes are nonbiodegradable materials such as polyethylene and plastic sheets, bags, and bottles. These materials clog soil pores and hamper natural drainage, tillage, and planting operations.

Municipal wastes may contain nonbiodegradable organics and heavy metals. There are a number of different ways in which cities dispose of their waste. The first and most well known, however, are dumps. These are disposal on open ground. More commonly used today to protect the environment, however, are landfills. These are areas that are specially created so waste can be put into the ground with little or no harm to the natural environment through pollution. Today, landfills

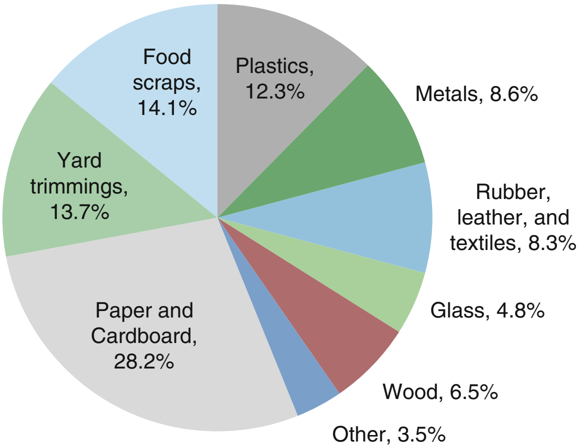


Fig. 6.1 Average composition of municipal solid waste (http://css.snre.umich.edu/css_doc/CSS04-15.pdf)

are engineered to protect the environment and prevent pollutants from entering the soil and possibly polluting groundwater in one of two ways. The first of these is with the use of a clay liner to block pollutants from leaving the landfill. These are called sanitary landfills, while the second type is called a municipal solid waste landfill. These types of landfills use synthetic liners like plastic to separate the landfill's trash from the land below it.

Municipal solid wastes (MSW) include wastes such as durable goods, for example, tires and furniture; nondurable goods, for example, newspapers, plastic plates, containers and packaging, milk cartons, plastic wrap, yard waste, and food. This category of waste generally refers to common household waste, as well as office and retail wastes, but excludes industrial, hazardous, and construction wastes.

In 2009, 54.3 % of MSW generated in the USA was disposed of in 1,908 landfills and 11.9 % was disposed of through waste incineration with energy recovery. Combustion reduces waste to ash by about 75 % by weight for disposal in a landfill. Another 33.8 % of MSW was recovered for recycling or composting, diverting 82 million tons of material from landfills and incinerators. Recovered composting materials represent 25 % of all recovered materials (USEPA 2010). The average composition of MSW has been shown in Fig. 6.1.

6.2.1.1 Composting Municipal Wastes

Composting is a biological process of reducing and stabilizing organic wastes through microbial digestion and decomposition into a complex of humic substances containing organic soil conditioners and plant nutrients. A variety of organic residues including municipal wastes are composted on both small farm scales and large commercial scales. Municipal wastes are generally composted at a central composting facility. During the process, part of organic C is released as CO_2, part incorporated into microbial cells and part humified. The organic nitrogen primarily as protein prior to composting is mineralized to inorganic N (NH_4-N and NO_3-N), which is then resynthesized into other forms of organic N in microbial biomass and humic substances during the composting process. In some composting programs, municipal wastes are separated at source into organic compostables and non-compostables by participating residents and businesses, while others accept a mixed stream and separate non-compostables at a centralized facility. However, some separation is needed even when organic compostables are separately collected. Composting programs which accept a mixed waste stream accept material more or less as it is collected, relying on the facility separation techniques described below. One important modification to traditional collection techniques is the establishment of a household hazardous waste (HHW) collection program. To be effective at reducing the contaminants of concern in MSW compost, such programs must emphasize heavy metal sources such as batteries and consumer electronics. The remaining mixed waste (less any separately collected recyclables and HHW) then serves as feedstock for the composting facility, where centralized separation of non-compostable materials will occur. With this approach, 60–70 % of the solid waste stream is typically processed into compost. The remaining 30–40 % includes recyclables as well as rejects destined for the landfill or an incinerator and landfill. The separating techniques are shown below.

Technology	Materials targeted
Screening	Large: film plastics, large paper, cardboard misc
	Midsized: recyclables, organics, misc
	Fines: organics, metal fragments, misc
Handpicking	Recyclables, inerts, and chemical contaminants
Magnetic separation	Ferrous plus contaminants associated with ferrous metal
Eddy current *Separation*	Nonferrous metals
Air classification	Lights: paper, plastic
	Heavies: metals, glass, organics
Wet separation	Floats: organics, misc
	Sinks: metals, glass, gravel, misc
Ballistic separation	Light: plastic, undecomposed paper
	Medium: compost
	Heavy: metals, glass, gravel, misc

After separation, the materials are reduced in volume and thoroughly mixed before delivering to the central composting facility. The composting is largely a biological process by which complex materials are decomposed into simpler substances with substantial mineralization, release of available nutrients, and further complexation. Proper design and management must be based on the needs of microorganisms if the process is to be a success. Composting is defined by human intervention into the natural process of decomposition. With a combination of proper environmental conditions and adequate time, microorganisms turn raw putrescible organic matter into a stabilized product. Through composting, readily available nutrient and energy sources are transformed into carbon dioxide, water, and a complex form of organic matter – compost. Process management can be optimized for a number of criteria, including the rate of decomposition (to reduce residence time in reactors and thus minimize facility size requirements), pathogen control, and odor management. The key parameters are the available carbon to nitrogen (C:N) ratio, moisture, oxygen, and temperature.

6.2.1.2 Benefits of Municipal Waste Compost

The bioconversion process is gradually emerging as a natural, promising, environment-friendly and potential microbial process to degrade environmental contaminants (Colwell 1994). At present, the municipal solid waste composting is being encouraged in many countries of the world and researchers have experienced the benefits of using MSW compost in the field (Paul and Howard 1997; Abigail 1998; Pokhrel and Viraraghavan 2005). However, the influence of C-rich materials, like municipal organic wastes compost, on soil physical, chemical, and biological properties depends upon several factors: amount and components of added organic materials, soil type, and weather conditions (Unsal and Ok 2001; Drozd 2003). As pointed out by Giusquiani et al. (1995), the use of composts from MSW improves the restoration of degraded soils and allows an appropriate final disposition of such materials, solving a major environmental and economical problem generated in the cities. Municipal waste composts have been used successfully for conditioning soil and supplying plant nutrients. Compost has the potential of being a fertilizer (Montemurro et al. 2005) and can thus be environmentally beneficial by substituting artificial fertilizers. To what degree the compost will enhance the nutrient status of the soil depends on both the waste that the compost is made from and the treatment technology (Guster et al. 2005). Even if the compost is low in nutrients, it can be valuable since application of compost, or any other form of humic material, effectively enhances soil structure, improves the water holding capacity, and reduces the sensitivity to erosion (Manser and Keeling 1996). Several tests also show that application of compost represses plant diseases in the field (Ros et al. 2005).

6.2.1.3 Municipal Waste Incineration

The role of waste incineration differs in the countries of the world. While in the industrialized countries in Europe as well as in Japan, the USA, and Canada the proportion of waste burned in waste incineration plants can be very high (up to 100 %), in most developing countries landfilling is the more common waste management practice.

The thermal treatment of solid municipal waste mostly takes place in plants equipped with grate firing systems; in individual cases, in pyrolysis, gasification, or fluidized bed plants; or in plants using a combination of these process stages. Residual municipal waste is delivered to grate furnaces as a heterogeneous mixture of wastes. Combustible components account for a content of about 40–60 %. Since the municipal waste incinerated is a heterogeneous mixture of wastes, in terms of sources of CO_2, a distinction is drawn between carbon of biogenic and carbon of fossil origin. The waste's carbon content is generally in the range of 28–40 %. Treatment in incineration plants is an output-controlled process. The combustion temperature of the gases in the combustion chamber as measured for at least two seconds after the last injection of combustion air is usually at least 850 °C. The oxygen necessary for incineration is supplied via ambient air, as primary, secondary, and/or tertiary air. At almost all municipal waste incineration plants, the heat produced during incineration is utilized for steam generation.

Hazardous waste is treated almost exclusively by incineration. Incineration must be understood here as an element of comprehensive logistics for the treatment of those wastes which due to their harmful nature have to be managed separately from municipal waste. Hazardous waste is waste requiring particular supervision, which by its nature, condition, or amount poses a particular hazard to health, air, and/or water or is particularly explosive or may contain or bring forth pathogens of communicable diseases. Since hazardous waste is generated for the most part in industrial production, notably the chemical industry, it is also referred to as industrial waste or industrial residue. Hazardous wastes occur, for example, as residues from petrochemical distillation processes, as undesirable by-products of syntheses processes of the basic organic chemical industry and the pharmaceutical industry, as well as in the recovery and disposal of contaminated or post-expiration-date products such as solvents, paints, or waste oil. In addition, environmental protection measures such as regulations prohibiting PCBs, CFCs, or halons may generate streams of hazardous waste. With combustion temperatures between 800 and 1,200 °C, the residence time of solids in the rotary kiln is up to 1 h, while for the combustion gases, it is only a few seconds. The waste gas generated during the combustion process is fed to an after burning chamber, in which the minimum temperature of between 850 and 1,200 °C is maintained for a residence time of at least 2 s.

6.2.1.4 Emissions from Incinerators

The incineration of municipal waste involves the generation of climate-relevant emissions. These are mainly emissions of CO_2 (carbon dioxide) as well as N_2O (nitrous oxide), NO_x (oxides of nitrogen), NH_3 (ammonia), and organic C, measured as total carbon. CH_4 (methane) is not generated in waste incineration during normal operation. It only arises in particular, exceptional cases and to a small extent (from waste remaining in the waste bunker), so that in quantitative terms CH_4 is not to be regarded as climate relevant. CO_2 constitutes the chief climate-relevant emission of waste incineration and is considerably higher, by not less than 10^2, than the other emissions.

The incineration of 1 Mg of municipal waste in MSW incinerators is associated with the production/release of about 0.7–1.2 Mg of carbon dioxide (CO_2 output). The proportion of carbon of biogenic origin is usually in the range of 33–50 %. The climate-relevant CO_2 emissions from waste incineration are determined by the proportion of waste whose carbon compounds are assumed to be of fossil origin.

The allocation to fossil or biogenic carbon has a crucial influence on the calculated amounts of climate-relevant CO_2 emissions. Other gases from incinerators include N_2O, CH_4, NO_x, CO, and NH_3.

6.2.1.5 Disposal of Incinerator Ash

Bottom ash is mainly glass and ceramic-like materials, also containing heavy metals and some organic carbon material. Dioxins were measured in the range 0.64–23 ngTEQ kg^{-1} (Friends of the Earth 2002). Bottom ash contains levels of dioxin similar to those found in urban soil. During the 5 years of the study period, 79 % of bottom ash from the Sheffield incinerator went to landfill and 21 % for bulk fill or substitute aggregate in construction blocks or asphalt. By 2000, 42 % of bottom ash was being processed. The report points out that bottom ash is used for engineering purposes in the Netherlands (100 % of ash), Denmark (70 %), France (50 %), and Germany (50 %). At bottom ash processing facilities, the ash is weathered to absorb water and reduce alkalinity. Oversize objects are removed, and then it is screened for size. Thirty-eight percent is used for bulk fill, road construction or asphalt, and 8 % for construction blocks. Thirty-eight percent goes to landfill and most of the rest into stockpile. The report estimated that dioxin levels in blocks made from bottom ash would be around 4 ng TEQ kg^{-1}. As this is comparable to levels found in soil, dust from drilling these blocks should be no more hazardous than dust from soil. Therefore, the report (Environment Agency Report; Friends of the Earth 2002) concludes that the use of bottom ash for aggregate is safe.

6.2.1.6 Municipal Solid Waste Landfill

A municipal solid waste landfill unit is a discrete area of land or an excavation that receives household waste and that is not a land application unit, surface impoundment, injection well, or waste pile. The municipal solid waste types potentially accepted by MSW landfills include (most landfills accept only a few of the following categories) MSW, household hazardous waste, municipal sludge, municipal waste combustion ash, infectious waste, waste tires, industrial nonhazardous waste, construction and demolition waste, agricultural wastes, oil and gas wastes, and mining wastes. In the USA, approximately 57 % of solid waste is landfilled, 16 % is incinerated, and 27 % is recycled or composted (USEPA 1992a). There were an estimated 2,500 active MSW landfills in the USA in 1995. These landfills were estimated to receive 189 million Mg of waste annually, with 55–60 % reported as household waste and 35–45 % reported as commercial waste.

Landfills are dangerous places; there are continuous release of volatile gases and leakage in the form of leachates. The chief gas emissions from landfills include methane (CH_4) and CO_2 and are produced by microorganisms within the landfill under anaerobic conditions (USEPA 1991). The environmental threats from two landfill areas of the USA, Volney and Colesville, show the danger of landfills. These landfills were old and unlined. The groundwater in the perimeter of the site contains volatile organic compounds (VOCs) (i.e., benzene, toluene, and chlorinated hydrocarbons) and heavy metals (i.e., arsenic, barium, cadmium, chromium, mercury, and nickel). Leachate collected from the landfill contains VOCs and heavy metals. Potential pathways of exposure to these contaminants included drinking contaminated groundwater and surface water, as well as accidental ingestion of contaminated soil and sediments. Modern landfills are lined with inert linings, but this does not exclude the danger. These materials get damaged in the long run.

6.2.1.7 Open Dumping of Municipal Wastes

In most of the cities of South Asia, open dumping is the most preferred method for the final disposal of solid waste. Even though government and municipalities are already working to develop the sanitary landfill sites in few urban areas, open dumping still remains the cheapest and most effective solution to get rid of the mounting garbage. These open dumpsites in the course of time become haven for scavengers (birds, animals, and human). Open dumping is the simplest and inexpensive method on trash disposal, but at the same time, it the easiest way of polluting air, water, and soil. Wastes are directly disposed of in water bodies or land. These wastes are usually mixed wastes which contain several organic, biodegradable, and persistent organic pollutants and heavy metals.

The chief disadvantages of open dumping are:

- Open dumping spreads foul odor.
- Open dumping provides the area for germs, viruses, fungi, bacteria, and vermin to breed, so it becomes the source of many diseases.
- Open dumping can contaminate the groundwater with inorganic and organic pollutants.
- Open dumping can contaminate the soil by leaking organic and heavy metal pollutants.
- Open dumping is prone to cause slide down.
- Open dumping wastes occupy an unnecessarily large area.

6.2.2 Sewage Sludge

Sewage sludge is usually a liquid mixture, composed both of solids and of dissolved organic and inorganic materials. The water is separated from the solid part by a number of treatments before it is environmentally safe for discharge into streams or lakes. The solid residue left is often discharged on open dumps, landfills, incinerators, or composted for agricultural use.

Most wastewater treatment processes produce sludge. Conventional sewage treatment plants typically generate a primary sludge in the primary sedimentation stage of treatment and a secondary, biological sludge in final sedimentation after the biological process. The characteristics of the secondary sludge vary with the type of biological process, and, often, it is mixed with primary sludge before treatment and disposal. Land application of raw or treated sewage sludge can reduce significantly the sludge disposal cost component of sewage treatment as well as providing a large part of the nitrogen and phosphorus requirements of many crops. But it has the risk of polluting the environment. Urban sewerage systems transport domestic sewage, industrial effluents, and storm-water runoff from roads and other paved areas. Thus, sewage sludge will contain organic waste material and traces of many pollutants used in our modern society. These substances can be phytotoxic and some toxic to humans (Dean and Suess 1985). Sewage sludge also contains pathogenic bacteria, viruses, and protozoa along with other parasitic helminths which can give rise to potential hazards to the health of humans, animals, and plants. A WHO (1981) report on the risk to health of microbes in sewage sludge applied to land identified salmonellae and *Taenia* as giving rise to greatest concern. The numbers of pathogenic and parasitic organisms in sludge can be significantly reduced before application to the land by appropriate sludge treatment. Sewage sludge contains useful concentrations of nitrogen, phosphorus, and organic matter. The availability of the phosphorus content in the year of application is about 50 % and is independent of any prior sludge treatment. The organic matter in sludge can improve the water retaining capacity and structure of some soils, especially when applied in the form of dewatered sludge cake. Sewage sludge should be subjected to biological, chemical, or thermal treatment, long-term storage, or other appropriate processes designed to reduce its fermentability and health hazards resulting from its use before being applied in agriculture.

Compost made from lime dewatered sludge tends to have a high pH. When blended with organic materials such as peat moss or soil, the resulting pH is generally between 7.2 and 8.0. Compost made from polymer dewatered sludge tends to have a pH between 6.2 and 6.8. When blended with peat moss or pine bark, the resulting pH is between 4.7 and 6.2, depending on the pH of the peat moss. Compost made from processed garbage and polymer dewatered sewage sludge (municipal compost) has a pH between 6.9 and 7.2 and also tends to have a high boron depending on the bulking material. These composts are advised to apply in landscape plantings, for shade trees, individually planted trees, flower gardens, etc. Sewage sludge is rich in organic matter, nitrogen, phosphorus, calcium, magnesium, sulfur, and other microelements necessary for plants and soil fauna to live. So it is characterized by the large manurial and soil-forming value. Except the indispensable elements to live, sludge can contain toxic compounds (heavy metals, pesticides) and pathogenic organisms (bacteria, eggs of parasites) (Siuta 1999).

The chemical composition of sewage sludge is of great importance for developing recommendations for the rates of sludge applications on agricultural land (Beltran et al. 1999). At the present time, recommendations for sludge applications rates on land are based on the fertilizer values (N, P, and K) and on the concentrations of trace metals present in sludge (Delgado et al. 1999). The metals of primary concern are Zn, Cu, Pb, Ni, and Cd which, when applied to soils in excessive amounts, may reduce plant yields or impair the quality of food or fiber produced (Parr et al. 1989).

Table 6.1 Acceptable amounts of heavy metals in sewage sludge for agricultural use (Chefetz et al. 1996; Siuta 1999; Moreno et al. 1999)

	Soil fertilization, land reclamation	
Metals	Agricultural exploitation	Nonagricultural exploitation
Lead (Pb)	500	1,000
Cadmium (Cd)	10	25
Chromium (Cr)	500	1,000
Copper (Cu)	800	1,200
Nickel (Ni)	100	200
Mercury (Hg)	5	10
Zinc (Zn)	2,500	3,500

On the other hand, repeated applications of compost from sewage sludge on agricultural soil have significant effects on the physical and chemical properties of soil (Reddy and Overcash 1981; Okereke 1985). A considerable amount of research has been carried out in Europe in order to evaluate the effects of compost from sewage sludge on soil properties (Williams and Goh 1982). The acceptable limits of some heavy metals in sludge for agricultural use are given in Table 6.1.

It has been shown in some studies that through application of appropriate rates of sewage sludge, plant growth and physical properties of soil improve substantially and their reusable nutritional values increase (Reed et al. 1991). The addition of sewage sludge gives a slight increase in Cu content of soils and Zn content of plants, which suggests that application of sewage sludge on agricultural lands should be carried out without surpassing the toxic limits set by the relevant organizations (Saruhan et al. 2010). Since sewage sludge contains high concentrations of potentially toxic elements such as Zn, Ni, Cd, and Cu, problems may arise when sludge is applied to an agricultural soil (Sanchez-Monedore et al. 2004; Madyiwa et al. 2002) and heavy metal accumulation also occurs in the plant tissues.

Biosolids, the product generated from tertiary treatment of waste activated sludge, and composted sludge have been used in agricultural fields of Europe and China for about a century. Substantial increase in crop yield due to sewage sludge application has been observed. But for heavy metal contamination of soil and public outrage, Austria, Switzerland, Sweden, and some other European countries have imposed a ban on the use of sewage sludge as a fertilizer.

6.2.3 Contaminants in Garden Composts

Contaminants in feedstock can impact the quality, marketability, and use of finished composts. Overuse and persistence of some insecticides could result in contamination of yard waste and compost. Since composting is a biologically mediated process, there is a considerable interest in knowing if the intense biological activity during the active thermophilic degradation period could induce degradation of pesticide contaminants. In the review of pesticide occurrence and degradation during composting, Buyuksonmez et al. (1999) found that a few of the targeted pesticides were found in the composts with organochlorine compounds being the most resistant to degradation during composting.

Some herbicides are resistant to degradation. Clopyralid and picloram had been detected in some compost (Bezdicek et al. 2001). Compost contaminated with clopyralid, a broadleaf herbicide, caused plant damage in Washington State in 1999, even though the damage largely disappears if the use of contaminated clippings as feedstock is delayed for a year or longer (Miltner et al. 2003). Residues from other broadleaf herbicides for lawn care, 2,4-D (2,4-dichlorophenoxy acetic acid), dicamba (2-methoxy-3,6-dichlorobenzoic acid), and MCPP [2-(4-chloro-2-methylphenoxy propionic acid)], were detected and found toxic to tomato (*Lycopersium esculentum* L.) (Bugbee and Saraceno 1994). Other pesticides that have also been detected in composts include atrazine, carbaryl, and chlordane. While diazinon [*O*,*O*-diethyl (*O*-2-isopropyl-6-methyl-4-pyrimidinyl)] insecticide was used extensively for insect control on turfgrass, its biodegradability is evident as only a trace of it (<1 %) could be found after composting (Michel et al. 1997). Close to 11 % of the insecticide was degraded during composting, and a majority of the insecticide was converted to a potentially leachable but less toxic hydrolyzed product. The release of diazinon through volatilization is extremely small (<0.2 %).

6.2.4 Hospital Wastes

A huge quantity of waste is generated in health-care facilities. These wastes include general wastes, sharps, non-sharps, blood, body parts, chemicals, pharmaceuticals, medical devices, and radioactive materials. Potentially infectious waste includes all waste items that are contaminated with or suspected of being contaminated with body fluids. Examples include blood and blood products, used catheters and gloves, cultures and stocks of infectious agents, waste from dialysis and dentistry units, waste from isolation units, wound dressings, nappies, discarded diagnostic samples, and contaminated materials (swabs, bandages, and gauze) and equipment (disposable medical devices). Wastes containing chemical substances such as laboratory chemicals, empty bottles of lab or pharmacy chemicals, disinfectants that have expired or are no longer needed, solvents, diagnostic kits, poisonous and corrosive materials, and cleaning agents. Genotoxic waste consists of highly hazardous, mutagenic, teratogenic, or carcinogenic waste containing substances with genotoxic properties. Radioactive hospital wastes include unused liquids from radiotherapy or laboratory research; contaminated glassware, packages, or absorbent paper; urine and excreta from patients treated or tested with radioactive substances. These wastes are often disposed with municipal wastes in open dumps and on lands.

6.2.5 Industrial Wastes

Industrial solid wastes are the major sources of soil pollution by toxic organic and inorganic chemical compounds and heavy metals. Industrial wastes include scrap metals, trash, oil, solvents, chemicals, cafeteria garbage, dirt and gravel, masonry and concrete, wood and scrap lumber, and similar wastes. Container wastes with solid, liquid, or gases are divided into hazardous and nonhazardous waste. Certain commercial products such as cleaning fluids, paints, or pesticides can also be defined as hazardous waste. Industrial waste may be toxic, ignitable, corrosive, or reactive. If improperly managed, or disposed, this waste can pose dangerous health and environmental consequences. In the USA, the amount of hazardous waste generated by manufacturing industries increased from an estimated 4.5 million tons annually after World War II to some 57 million tons by 1975. By 1990, this total had shot up to approximately 265 million tons. This waste is generated at every stage in the production process. The introduction of many new products for homes and offices – computers, drugs, textiles, paints and dyes, and plastics – also introduced hazardous waste, including toxic chemicals, into the environment. The EPA estimated in 1980 that more than 70,000 different chemicals were being manufactured in the USA with some 1,000 new chemicals being added each year. The human health and environmental impacts of many of these chemicals are largely unknown. High levels of toxic contaminants have been found in animals and humans, particularly those, like farm workers and oil and gas workers, who are continually exposed to such waste streams. Wastewater from manufacturing or chemical processes in industries contributes to soil and water pollution. Most major industries have treatment facilities for industrial effluents but this is not the case with small-scale industries, or in industries of the undeveloped countries, which cannot afford enormous investments in pollution control equipment as their profit margin is very slender. Polluted water is unsuitable for drinking, agriculture, recreation, and industry. Two types of industrial wastes of particular concern are dry cleaning fluids and embalming fluids. Dry cleaning fluids have contaminated groundwater supplies in all areas in the USA. One of the most common contaminants is PCE (perchloroethylene or tetrachloroethylene $Cl_2C{=}CCl_2$). Embalming fluids include carcinogens. The threat of embalming fluids contaminating the water supply has prompted some cities to decrease the size of planned large cemeteries (CEQ 1981) (Table 6.2).

6.2.6 Agrochemicals

Agrochemicals include various chemical substances used for production and protection in agriculture. In most cases, it refers to the broad range of pesticides, including insecticides, herbicides, and fungicides. It may also include synthetic fertilizers, hormones and other chemical growth agents, and concentrated stores of raw animal manure. Most agrochemicals are toxic, and their bulk storage may pose significant environmental and/or health risks, particularly in the event of accidental spills.

Table 6.2 Hazardous waste generated by industries and businesses

Industries/manufacturer	Waste type
Chemical manufacturers	Acids and bases
	Spent solvents
	Reactive waste
	Wastewater containing organic constituents
Printing industry	Heavy metal solutions
	Waste inks
	Solvents
	Ink sludges containing heavy metals
Petroleum refining industry	Wastewater containing benzene and other hydrocarbon
	Sludge from refining process
Leather products manufacturing	Toluene and benzene
Paper industry	Paint waste containing heavy metals
	Ignitable solvents
Construction industry	Ignitable paint waste
	Spent solvent
	Strong acids and bases
Metal manufacturing	Sludges containing heavy metals
	Cyanide waste
	Paint waste

In many countries, use of agrichemicals is highly regulated. Fertilizers and manures cause soil pollution by introducing heavy metals and their compounds into the soil. The heavy metals associated with some phosphate fertilizers are arsenic, cadmium, manganese, uranium, vanadium, and zinc. Common phosphate fertilizers are manufactured from rock phosphates, and the heavy metal contents of the raw material depend on their source ores (Helal and Sauerbeck 1984). Poultry manure and pig manure may pollute the soil with zinc, copper, and arsenic.

A pest is a living organism that degrades the health, value, utility, condition, or amenity of another organism, a structure, or a place. Pest organisms can be plants, fungi, algae, vertebrate or invertebrate animals, or microorganisms such as bacteria, molds, slimes, and fungi. Pesticides are substances or organisms used to eliminate, incapacitate, modify, inhibit growth of, or repel pests. They can be natural or synthetic chemicals, mixtures of these, or living organisms that act as biological control agents. According to the British Food and Environment Protection Act, 1985, a pesticide is defined as any substance or preparation prepared or used for any of the following purposes:

- Destroying organisms harmful to plants or to wood or other plant products
- Destroying undesired plants
- Destroying harmful creatures

Pesticides are divided into insecticides, fungicides, and herbicides according to their target organisms. The use of pesticides in agriculture has been steadily increasing during the recent decades.

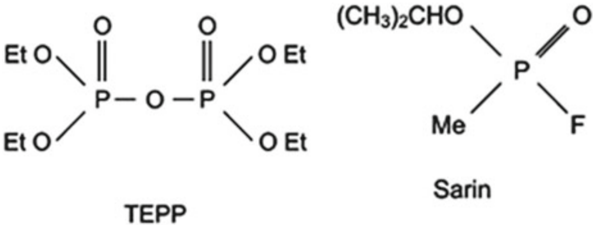


Fig. 6.2 Two organophosphorus pesticides

Pesticides applied to plants or harmful organisms (plant, microorganism, insects) undergo repeated dissolution, adsorption, and elution with the soil solution and colloidal phases and spread to all directions within soil, water, and atmosphere systems. This mobility depends on solubility, adsorbability, and volatility.

6.2.6.1 Insecticides

The use of insecticides has been increasing after the end of World War II. There are a great number of substances classified as insecticides, but they can belong to any of the four groups of organic compounds: the organophosphorus compounds, the organochlorines, the carbamates, and the pyrethroids.

Organophosphorus Compounds

Organophosphorus compounds are technically nerve poisons because they act on the central nervous systems. Developed during World War II, they are used in many different ways in agriculture. They can be used as fumigants, some as contact poisons, and others as systemic pesticides. The most important members of the group are tetraethyl pyrophosphate (TEPP) and sarin (Fig. 6.2). Sarin is also used in warfare. Many organophosphates are highly toxic to nontarget organisms.

Organophosphates manufactured latter are less toxic to mammals but toxic to target organisms, such as insects. Malathion, Dibrom, chlorpyrifos, temephos, diazinon, and terbufos are organophosphates.

Organochlorines

Some organochlorine compounds, also developed during World War II, were found to be very effective in controlling pests responsible for diseases such as malaria and yellow fever. These compounds were cheap and easily available, and they were thought to be safe for humans. They belonged to the three families: the DDT (dichlorodiphenyltrichloroethane) family, the BHC family, and the cyclodiene family (Fig. 6.3). The principal representative of the BHC family is called lindane. Aldrin, dieldrin, and heptachlor are stereochemically related compounds and belong to the cyclodiene

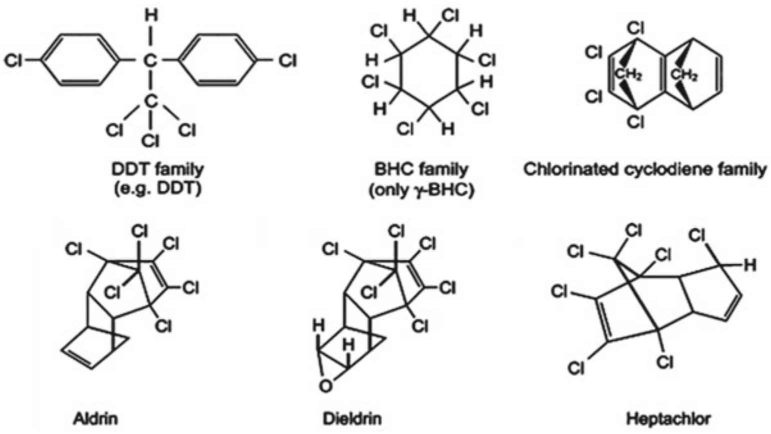


Fig. 6.3 Some members of the organochlorine pesticides

family, which were effectively used for locusts. Organochlorines have been effectively used in the past in agriculture and hygiene, but they have been found latter to persist in the environment and kill beneficial and harmless organisms, as well.

Carbamates

Carbamates are organic compounds derived from carbamic acid (NH_2COOH). A carbamate group, carbamate ester (e.g., ethyl carbamate), and carbamic acids are functional groups that are interrelated structurally and often are interconverted chemically. Carbamate esters are also called urethanes. Aldicarb, carbaryl, propoxur, oxamyl, and terbucarb are carbamates. Although these pesticides differ chemically, they act similarly. When applied to crops or directly to the soil as systemic insecticides, carbamates generally persist from only a few hours to several months. However, they have been fatal to large numbers of birds on turf and in agriculture and negatively impacted breeding success in birds. These insecticides kill insects by reversibly inactivating the enzyme acetylcholinesterase. The organophosphate pesticides also inhibit this enzyme, although irreversibly, and cause a more severe form of cholinergic poisoning. Figure 6.4 shows structures of some carbamate insecticides.

Pyrethroid Insecticides

Natural pyrethroids are extracted from dried pyrethrum or chrysanthemum flowers. Pyrethroids comprise of six active ingredients: pyrethrum I and II, cinerins I and II, and jasmolines I and II (Fig. 6.5). Synthetic pyrethroids have higher insecticidal activity. Some pyrethroids are preferred for their lower persistence and lower toxicity. The four groups of synthetic pyrethroids are as follows: allethrin, bioresmethrin,

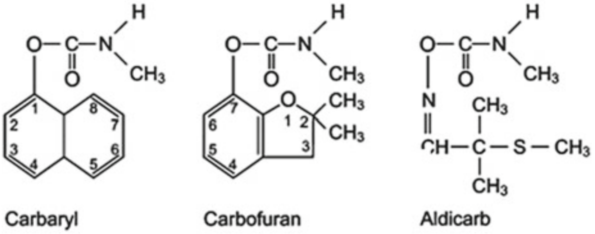


Fig. 6.4 Carbamate pesticides

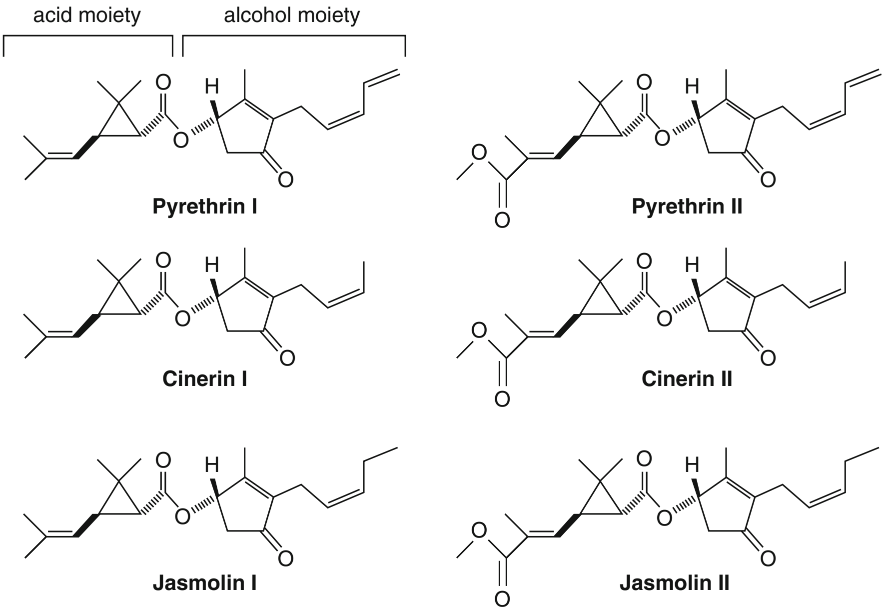


Fig. 6.5 Molecular structures of six natural pyrethrins

permethrin, and fenvalerate groups. Pyrethroids are used in many household insecticides. They provide quick "knockdown" action and rapidly degrade in heat and sunlight. This eliminates their usefulness in agricultural pest control. Pyrethroids are relatively nontoxic to mammals and birds. However, they are highly toxic to aquatic organisms and bees.

6.2.6.2 Herbicides

Herbicides are chemical substances used to control weeds. It has become an essential management practice in conservation tillage systems, because no till and minimum tillage cannot eradicate weeds. A few hundreds of organic and inorganic substances of more than 50 chemical groups are used as herbicides. The following are some

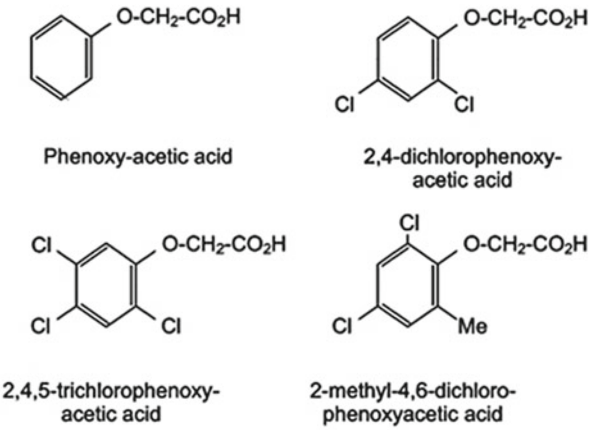


Fig. 6.6 Organochlorine herbicides

Phenoxy-acetic acid

2,4-dichlorophenoxy-acetic acid

2,4,5-trichlorophenoxy-acetic acid

2-methyl-4,6-dichloro-phenoxyacetic acid

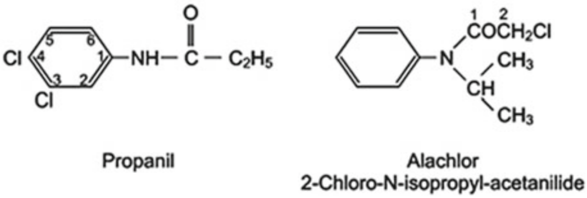


Fig. 6.7 Aniline derivatives used as herbicides

Propanil

Alachlor
2-Chloro-N-isopropyl-acetanilide

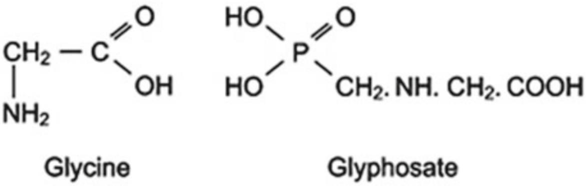


Fig. 6.8 Structures of glycine and glyphosates

Glycine

Glyphosate

popular examples. A classification of herbicides may be obtained from the web page http://www.alanwood.net/pesticides/class_herbicides.html.

Organochlorine compounds contain derivatives of phenoxyacetic acid, such as 2,4-dichlorophenoxyacetic acid known as 2,4-D and 2,4,5-trichlorophenoxyacetic acid 2,4,5-T or 2-methyl-4,6-dichlorophenoxyacetic acid, known as MPCA (Fig. 6.6). An equal mixture of 2,4-D and 2,4,5-T is known in the code name Agent Orange. The US Army sprayed millions of hectares of woodlands during their war against Vietnam. Vietnamese still suffer from damages caused by the substance. Besides derivatives of phenoxyacetic acid, there are derivatives of aniline among organochlorine herbicides. Two aniline derivatives are propanil and alachlor (Fig. 6.7).

Organophosphorus herbicides (known as glyphosates) are widely used in agriculture for their weed killing efficiency and noncarcinogenic nature. Glyphosate is a modified glycine (Fig. 6.8).

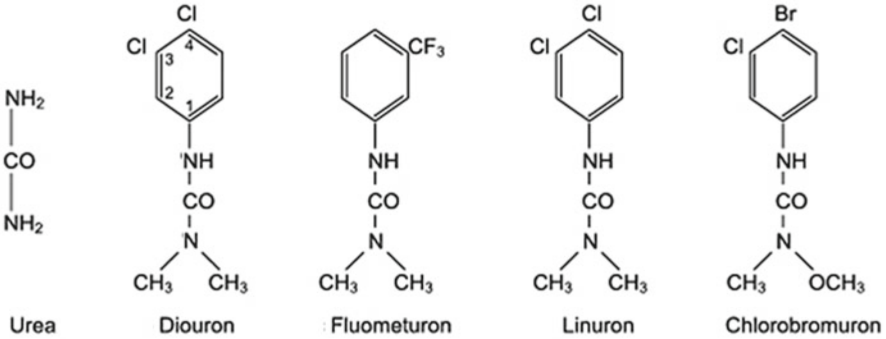
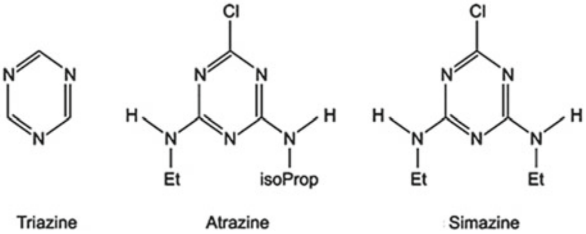


Fig. 6.9 Some urea derivatives used as herbicides

Fig. 6.10 Structures of some triazine derivatives



Derivatives of carbamic acid including several urea derivatives (Fig. 6.9) are also used as herbicides.

The triazine family of herbicides (Fig. 6.10), which includes atrazine, was introduced in the 1950s; they have the current distinction of being the herbicide family of the greatest concern regarding groundwater contamination. Atrazine is persistent for some week after being applied to soils of above-neutral pH. Although banned in Europe, triazines are still widely used in many countries to control weeds in forest plantations and in crops such as canola, sugar cane, maize, sorghum, and lupins.

6.2.6.3 Fungicides

Fungicides are chemical compounds, inorganic and organic, natural and synthesized, which are used to control fungi. These are used to control pathogenic diseases of plants, to protect seeds and grains during storage, as fumigants, and the like. Some examples of fungicides are given below.

- Inorganic and organic compounds of heavy metals. Examples include mixtures of copper-bearing inorganic compounds (e.g., Bordeaux mixtures) or organometallic compounds such as organotins (tributyltinacetate or triphenyltin acetate) (Fig. 6.11).
- Derivatives of phthalic acid. One example is phthalimide. The most familiar name is captan (Fig. 6.12).

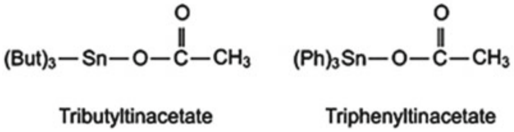


Fig. 6.11 Two organotin fungicides

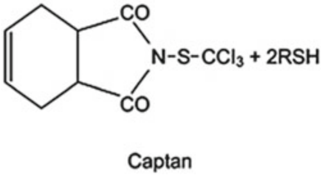


Fig. 6.12 Structure of phthalimide

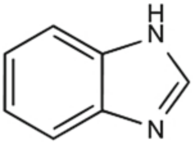


Fig. 6.13 Structure of benzimidazole

- Benzimidazoles. Benzimidazole is a systemic fungicide. The pentagonal ring in histamine is known as an imidazole ring. Its fusion with a benzene nucleus gives the benzimidazole (Fig. 6.13).

6.2.6.4 Persistence of Pesticides in Soil

Persistence may be defined as the tendency of a given compound, a pesticide in this case, to conserve its molecular integrity and chemical, physical and functional characteristics in a medium through which it is transported and distributed after being released into the environment. Many organic compounds persist for long periods in soils, subsoils, aquifers, surface waters, and aquatic sediment. The low- and high-molecular weight compounds that resist biodegradation are known as recalcitrant molecules. Many pesticides, mainly herbicides, have this characteristic (Navarro et al. 2007).

Organochlorine pesticides are considered persistent because they are stable in the environment and can resist degradation. This ability of organochlorine pesticides makes them highly effective and therefore widely used but at the same time helps them to build up their residues in soil and water. DDT was used in large scales to control typhus and malaria. The properties that made these chemicals such effective insecticides also made them environmental hazards; they accumulate in human fat tissue. Some of the organochlorine pesticides have been banned for use in US and European countries, but others are the active ingredients of some home and garden products and some agricultural and environmental pest-control products. There are four broad groups of organochlorine pesticides:

- Hexachlorocyclohexane (lindane)
- DDT and related compounds, DDE and DDD
- Cyclodienes (aldrin, heptachlor, and others)
- Mirex and chlordecone

Eleven specific persistent pesticides within these groups were measured during the Churchill County leukemia study (16 children diagnosed with acute leukemia between 1997 and 2002 lived in Churchill County, Nevada; the probable association of persistent pesticide residues could not, however, be proved; Rubin et al. 2007): beta-hexachlorocyclohexane, gamma-hexachlorocyclohexane (lindane), hexachlorobenzene, dieldrin, mirex, heptachlor epoxide, oxychlordane, *trans*-nonachlor, *o*,*p*,-DDT, *p*,*p*,-DDT, and *p*,*p*,-DDE.

Herbicides are applied to the soil in hopes of obtaining season-long weed control. It is desirable for the chemicals to control weeds during the season of application, but they should not remain long enough to affect subsequent crop growth. The length of time that a herbicide remains active in the soil is called "soil persistence" or "soil residual life." Anything that affects the disappearance or breakdown of a herbicide affects persistence. Many factors determine the length of time herbicides persist. Most factors fall into three categories: soil factors, climatic conditions, and herbicidal properties. These categories strongly interact with one another. Herbicides vary in their potential to persist in the soil. Some herbicide families that have persistent members include the triazines, uracils, phenylureas, sulfonylureas, dinitroanilines, pigment inhibitors, imidazolinones, and certain plant-growth regulators. Several common herbicides in these groups are shown below.

- *S-triazines*: atrazine (AAtrex, Atrazine), hexazinone (Velpar), prometon (Pramitol), and simazine (Princep)
- *Dinitroanilines*: benefin (Balan), oryzalin (Surflan), pendimethalin (Pendimax, Prowl), prodiamine (Barricade), and trifluralin (Treflan, Tri-4, TRILIN)
- *Others*: bensulide (Betasan, Prefar), clomazone (Command), and tebuthiuron (Spike)
- *Phenylureas*: diuron (Karmex, Direx)
- *Uracils*: bromacil (Hyvar X) and terbacil (Sinbar)
- *Imidazolinones*: imazapyr (Arsenal), imazaquin (Scepter), and imazethapyr (Pursuit)
- *Sulfonylureas*: chlorimuron (Classic), chlorsulfuron (Telar), nicosulfuron (Accent), primisulfuron (Beacon), prosulfuron (Peak), and sulfometuron (Oust)
- *Plant-growth regulators*: clopyralid (Stinger), picloram (Tordon), triclopyr (Garlon), and chlorimuron

Pesticide persistence in soil is not the property of the pesticides alone; it also depends on soils, climate, and management. At higher soil pH, persistence of some pesticides decreases. At lower pH, the persistence of clomazone and the imidazolinones (imazaquin and imazethapyr) decreases. Soil pH has little effect on the persistence of some other herbicides. Various nutrients and cations in the soil affect both herbicide activity and degradation. The CEC, principally a function of clay-type and organic matter content, is directly involved in herbicide adsorption. Some herbicides are more available in the presence of certain cations, whereas others may be tied up and therefore unavailable. The literature indicates that there is much variation in the effect that cations and nutrients can have on herbicide activity and breakdown, depending on soil composition, nutrient type and concentration, and chemistry of the herbicide.

Soil microorganisms are partially responsible for the breakdown of many herbicides. The types of microorganisms and their relative amounts determine how quickly decomposition occurs. Soil microbes require certain environmental conditions for optimal growth and utilization of any pesticide. Factors that affect microbial activity are temperature, pH, oxygen, and mineral nutrient supply. Usually, a warm, well-aerated, fertile soil with a medium soil pH is most favorable for pesticide degradation. The persistence of some pesticides in soil is shown below:

Persistence	Common herbicides
1 month	2,4-D, glufosinate, glyphosate, MPCA
1–3 months	Acetochlor, alachlor, bentazon, butylate, DCPA, dimethenamid, EPTC, flumetsulam, foramsulfuron, halosulfuron, lactofen, linuron, mesotrione, metolachlor, metribuzin, naptalam, siduron
3–12 months	Atrazine, benefin, bensulide, bromoxynil, clomazone, diuron, ethalfluralin, homesafen, hexazinone, imazaquin, imazethapyr, isoxaflutole, oryzalin, pendimethalin, primisulfuron, prodiamine, pronamide, prosulfuron, simazine, sulfentrazone, terbacil, topramezone, trifluralin
>12 months	Bromacil, chlorsulfuron, imazapyr, picloram, prometon, sulfometuron, tebuthiuron

6.2.6.5 Processes of Accumulation of Pesticide Residues in Soil

Pesticides applied on crops and pests undergo several transformations, including volatilization to the atmosphere, microbial assimilation, biochemical degradation, photochemical degradation, diffusion, erosion and runoff, absorption by plants, leaching to the groundwater, and accumulation in soil (Fig. 6.14). These processes are responsible for the movement of pesticides and their residues within the environmental components.

The principal process of pesticide accumulation in soil is adsorption, which may be chemical in nature (as with electrostatic interactions) or purely physical (as with van der Waals forces). Adsorption takes place between charged pesticide molecules (sorbate) and soil particles (adsorbent), including clay minerals, sesquioxides, and humus. Positively charged pesticide molecules can bind to negatively charged particles of clay and organic matter. The extent of adsorption depends on the properties of soil and the compound, which include size, shape, configuration, molecular structure, chemical functions, solubility, polarity, polarizability and charge distribution of interacting species, and the acid–base nature of the pesticide molecule (Senesi 1992; Pignatello and Xing 1996). Soil pH, or the acid/base balance of the soil solution, affects the chemical's reactivity and certain soil functions such as microbial metabolism. Weber et al. (1969) showed that maximum adsorption of basic compounds occurs

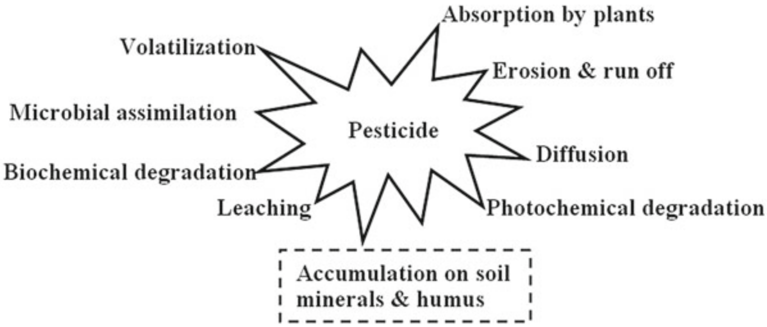


Fig. 6.14 Transformations of applied pesticides

at pH values to their pKa value. Acidic and anionic pesticides, such as phenoxyacetic acids (2,4-D and 2,4,5-T) and esters, asulam and dicamba, can interact with soil organic matter by H-bonding at pH values below their pKa in nonionized forms through their −COOH, −COOR, and identical groups (Senesi et al. 1984).

Chemical reactions between pesticides and their metabolites often lead to the formation of strong bonds (chemisorption), which increase in the persistence of the residues in the soil, but decrease their bioavailability and toxicity (Dec and Bollag 1997). It also reduces leaching and transport properties. The nature of the binding forces involved and the types of mechanisms operating in the adsorption processes of pesticides onto the soil include ionic, hydrogen, and covalent bonding; charge transfer or electron donor–acceptor mechanisms; van der Walls forces; ligand exchange; and hydrophobic bonding or partitioning (Gevao et al. 2000).

6.2.7 *Atmospheric Deposition*

Atmospheric deposition refers to substances that are deposited on land from the air. It includes gas, smoke, ashes, and particulates and deposited to soil. The oxides of sulfur and nitrogen (SO_x, NO_x), chlorides, fluorides, ammonium, etc. are the principal gas component of aerosols. Aerosols contain dust, smoke, ashes, sea salt, water droplets, and some organic materials. Volcanic eruption and burning of biomass and fossil fuel produce aerosol that contains sulfate compounds. The occurrence of aerosols is the highest in the northern hemisphere where industrial activity is concentrated. Suspended dust particles in air are carried to hundreds of kilometers by the wind and settle on soil. Atmospheric deposition may contain persistent organic pollutants as PCBs, and PAHs and heavy metals.

Atmospheric deposition occurs through three different processes: (1) wet deposition, the deposition of material with precipitation; (2) dry deposition, the direct deposition of atmospheric particles and gas to vegetation, soils, and surface water; and (3) cloud deposition, the deposition of nonprecipitating droplets of clouds and fogs to terrestrial surfaces (Fowler 1980; Lovett and Kinsman 1990). Wet and dry depositions may occur everywhere, while cloud deposition mainly occurs in coastal and mountainous due to their immergence in cloud and fog. Wet deposition occurs due to incorporation of particles and gases into cloud droplets and their subsequent precipitation as rain or snow. Wet deposition may also result from scavenging of particles and gases by raindrops and snowflakes as they fall. These processes are controlled by concentration and size distribution of particles and the solubility and reactivity of gases and by meteorological factors. There are such gases as oxides of nitrogen and sulfur dissolved in rainwater. In dry deposition, particles >5 μm diameter are deposited mainly by gravitational sedimentation and inertial impaction. Brownian diffusion is the process of dry deposition of particles ~2 μm (Fowler 1980). Particles between 0.2 and 2 μm are not readily dry deposited. They are deposited by wet deposition. The characteristics of the chemical species of interest in air pollutant dry deposition on soil and ecosystems have sufficiently been studied. The particle-associated substances derived primarily from erosion of soil containing K, Ca, Mg, Al, and Si tend to reside on larger airborne particles which can be deposited by gravitational sedimentation (Lindberg and McLaughlin 1986). In contrast, the majority of airborne mass of particulate SO_4^{2-}, NH_4^+, H^+, and Pb reside with sub-micrometer aerosol, which can be deposited by wet deposition. Nitrates reside on particles of wide ranges of sizes. Nitrate and SO_4^{2-} may occur on larger particles through adsorption of reactive gases like SO_2 and nitric acid vapor (HNO_3) on large alkaline or sea salt particles in the atmosphere (Butler 1988). Deposition in the form of dry particulate matter and wet precipitation can be acidic due to air pollution from burning fossil fuels. Sulfur dioxide (SO_2) and nitrogen oxides (NO, NO_2) react in the atmosphere to form sulfuric and nitric acids, respectively. These acids can alter the environment where they fall from the atmosphere, which can be long distances from the pollution source. The impacts of wet acidic deposition or acid rain are usually noticed in the higher elevations or ridgetops that receive more deposition due to the orographic effect and also in areas that have geological formations with limited buffering capacity (Kimmel 1999). Impacts to the environment include damage to trees, depletion of nutrients in the soil, and acidic stream water. The combination of low pH and toxic aluminum in water causes osmoregulatory failure and can cause the depletion of fish populations in headwater streams.

Heavy metals are emitted into the atmosphere through industrial, motor transport and volcanic activities, soil erosion, forest fires, evaporation, etc. (Salomons and Forstner 1984). Most of the metals in the air are in particulate form with the exception of mercury (Hg). Usually, toxic metals such as Pb, Cd, and As are found in fine particles (<2.5 μm). Fine particles are respirable and tend to persist in the atmosphere where they can undergo chemical reactions and be transported from their sources over long distances to pristine areas of the environment (Ross 1987). Most of heavy metals are soluble in water; thus, they can migrate in ecosystems by water pathways. Even small amounts and low concentrations of heavy metals can damage ecosystems and are dangerous for human health (Ovadnevaite et al. 2006). Luo (2009) reported atmospheric deposition of As, Cr, Hg, Ni, and Pd to soils.

The deposition fluxes of inorganic chemicals New Jersey precipitation is given in Table 6.3 for an understanding of the magnitude of atmospheric deposition.

Table 6.3 Deposition fluxes of heavy metals in New Jersey precipitation annual deposition fluxes (ng m^{-2} $year^{-1}$)

	Regions		
Element	New Brunswick	Jersey City	Pinelands
Ag	25	21	59
Al	35,000	28,000	24,000
As	67	160	65
Cd	62	76	23
Co	46	130	22
Cr	150	180	56
Cu	1,500	2,200	590
Fe	47,000	47,000	23,000
Hg	11	14	11
Mg	65,000	81,000	54,000
Mn	2,500	1,900	2,900
Ni	650	1,200	290
Pb	1,700	2,500	650
V	480	880	410
Zn	7,800	8,800	5,500

Source: Adapted from New Jersey Atmospheric Deposition Network (NJADN) Project data; (John et al. 2004) http://www.nj.gov/dep/dsr/trends/pdfs/atmospheric-dep-pcbs.pdf

The concentration, however, varies from place to place depending on the industrialization and emission, land use and erosion, urbanization, etc.; the North American territory is particularly known for acid rain.

6.2.8 Mining

Mining wastes include waste generated during the extraction, beneficiation, and processing of minerals. Most extraction and beneficiation wastes from hard rock mining (the mining of metallic ores and phosphate rock) and 20 specific mineral processing wastes are categorized by USEPA as “special wastes” and have been exempted by the Mining Waste Exclusion from federal hazardous waste regulations under Subtitle C of the Resource Conservation and Recovery Act. These are slag from primary copper processing, slag from primary lead processing, red and brown muds from bauxite refining, phosphogypsum from phosphoric acid production, slag from elemental phosphorous production, gasifier ash from coal gasification, process wastewater from coal gasification, calcium sulfate wastewater treatment plant sludge from primary copper processing, slag tailings from primary copper processing, fluorogypsum from hydrofluoric acid production, process wastewater from hydrofluoric acid production, air pollution control dust/sludge from iron blast furnaces and iron blast furnace slag, treated residue from roasting/leaching of chrome ore, process wastewater from primary magnesium processing by the anhydrous process,

process wastewater from phosphoric acid production, basic oxygen furnace and open hearth furnace air pollution control dust/sludge from carbon steel production, basic oxygen furnace and open hearth furnace slag from carbon steel production, chloride process waste solids from titanium tetrachloride production, and slag from primary zinc processing.

Six groups of wastes are classified hazardous: acid-generating tailings from processing of sulfide ore, other tailings containing dangerous substances, other wastes containing dangerous substances from physical and chemical processing of metalliferous minerals, wastes containing dangerous substances from physical and chemical processing of non-metalliferous minerals, oil-containing drilling muds and wastes, and drilling muds and other drilling wastes containing dangerous substances. The majority of hazardous waste (90 %) are produced by 18 high productivity industrial units (metallurgy, petroleum refining, chemical industry, fertilizer industry) while the rest of them (10 %) by about 500 smaller industries. There are two principal types of solid mine waste: waste rock and tailings.

6.2.8.1 Waste Rock

A mining operation needs to move and dispose of a large amount of blasted rock that does not have useful concentrations of minerals – this is called "waste rock." Waste rock is generated by both open pit mines and underground mines, but the volume is higher with open pit mines. The amount of waste rock, compared with the amount of ore is called the strip ratio. A strip ratio of 1 (volume of ore = volume of waste rock) is considered low for an open pit mine. The largest open pit gold mines have a strip ratio around 6. Waste rock is typically dumped into large piles within the mines waste rock storage area, which can spread over an area of several square kilometers. Both the physical and chemical characteristics of the waste rock must be considered if it is to be properly disposed of. The coarse texture of waste rock allows air and water to easily move through the pile. When exposed, waste rock can be very reactive with the air, water, and microorganisms and may cause acid mine drainage and release metals to surface and groundwater.

6.2.8.2 Tailings

Modern mines process huge quantities of ore, tens of thousands to hundreds of thousands of tons a day. After blasting and hauling from the mine shaft of pit, the ore is crushed and processed using massive volumes of water and a variety of chemical and physical processes. The mineral content of an ore can be in the 5 % range for base metals or as low as 0.00005 % for precious metals like gold. This means that 95–99.9995 % of the mined and processed ore becomes a waste product – known as tailings. Tailings are usually deposited as a slurry – a thick liquid made up of water, the finely ground ore, and any residual chemicals from the processing stages. Because the rock has been finely ground, tailings can be very chemically reactive and can pose serious environmental risks from acid rock drainage and the release of toxic metals, and toxic reagents used in processing. The combination of liquids and fine-grained solids make many tailings physically unstable. If left exposed to the air and dried, tailings can also be blown on the wind causing air pollution and washed into waterways and spread on land. Toxic pollutants that are commonly found in tailings include cyanide, mercury, copper, lead, arsenic, cadmium, selenium, zinc, and nickel.

6.2.8.3 Acid Mine Drainage

Acid mine drainage occurs when rocks with sulfur bearing minerals are exposed to air, water, and microorganisms. The sulfur in the minerals turns into sulfuric acid and moves in drainage water. Acid mine drainage is highly acidic and reacts with rocks and minerals of the dump. The chemical changes within the minerals also release significant amounts of toxic heavy metals like nickel, copper, and cadmium. These metals can contaminate surface soils and waters and find their way into food webs resulting in both acute and chronic impacts on wildlife and people.

6.2.9 Traffic

Pollution in soils from traffic sources occurs along major highways due to emission of particulate matter (de Kok et al. 2006), as well as gaseous pollutants, which can be transported to soil by both wet and dry deposition. Many of the substances released from traffic are insoluble in water, have high adsorption ability, and tend to bind to mineral and organic particle in soil. The pollutants can be retained or transformed in the soils or subsequently modified by soil microorganisms (Wesp et al. 2000).

Roadside soils often contain high concentrations of metallic contamination. The bioavailability and environmental mobility of the metals are dependent upon the form in which the metal is associated with the soil. Lead street dust and roadside soil has been extensively studied and found to be present at elevated levels (Goldsmith et al. 1976; Harrison et al. 1980). The lead in roadside soil is mainly found in the form of lead sulfate (Harrison et al. 1980). Metals such as Cu, Fe, Zn, and Cd are essential components of many alloys, wires, tires, and many industrial processes and could be released into the roadside soil and plants as a result of mechanical abrasion and normal wear. Analyses of roadside soil (Hewitt and Candy 1990) and plants (Otte et al. 1991) revealed that they contain elevated levels of these heavy metals.

A number of studies have investigated the release of pollutants from traffic into air (Klein et al. 2006). Combustion of fossil fuels in vehicle engines is an important source of a group of highly abundant pollutants called polycyclic aromatic hydrocarbons (PAHs). In addition, traffic can be a source of their derivatives, including some persistent organic pollutants (POPs). Some of these contaminants, such as polychlorinated biphenyls (PCBs) and polychlorinated dibenzo-*p*-dioxins and dibenzofurans

(PCDD/Fs) (Safe 1986), are hazardous because of their toxicity and persistence. POPs have nonpolar molecules and hence can accumulate in adipose tissue and cause deleterious cellular effects. The potential adverse effects of these compounds and their environmental mixtures include teratogenicity, carcinogenicity (Muto et al. 1996), and effects on normal physiologic endocrine function of an organism (Ankley et al. 1998). Human fertility has been suggested to be adversely affected by exposure to pollution from traffic (de Rosa et al. 2003). Some studies have demonstrated in vitro estrogenic as well as antiestrogenic and antiandrogenic effects of traffic exhaust particulates and road dust (Okamura et al. 2004; Misaki et al. 2008).

6.2.10 Radionuclides in Soil

Nuclides of any element that have atomic number greater than bismuth-83 are unstable and therefore radioactive (Igwe et al. 2005). They are called heavy nucleons or radionuclides. Radionuclides disintegrate or change spontaneously with a loss of energy in the form of ionizing radiation (van der Perk 2006). There are more than 60 radionuclides in nature. They can be placed in three general categories: primordial, cosmogenic, and anthropogenic. Examples of some naturally occurring radionuclides are americium-241, cesium-137, cobalt-60, iodine-129 and iodine-131, plutonium, radium, radon, strontium-90, technetium-99, tritium, thorium, and uranium. Cesium-137 is produced in nuclear fission, and it has a radioactive half-life of 30.17 years. Primordial radionuclides are left over from the creation of the earth. They typically have half-lives of hundreds of millions of years. Examples include uranium-235, uranium-238, thorium-232, and potassium-40. Primordial radionuclides end up in soil as part of the rock cycle.

Nuclear fission for atomic weapons testing and nuclear power generation provides some of the sources of soil contamination with anthropogenic radionuclides. To the naturally occurring radionuclides in soil such as ^{40}K, ^{87}Rb, ^{14}C, ^{235}U, ^{238}U, and ^{232}Th, a number of fission products have been added. However, only two of these are sufficiently long-lived to be of significance in soils: strontium-90 and cesium-137 with half-lives of 29.1 and 30 years, respectively. The average levels of these nuclides in soil in the USA are about 388 mc km^{-2} for 90Sr and 620 mc km^{-2} for 137Cs (Holmgren et al. 1993). These levels of the fission radionuclides in soil are not high enough to be hazardous (Igwe et al. 2005). However, artificial radioactivity may be released into the environment during the normal operations of nuclear facilities and installations such as nuclear ore processing, uranium enrichment, fuel fabrication, reactor operations, and application of radioisotopes in the fields of nuclear medicine, research, industry, and agriculture. Soils may receive some radionuclides from radioactive waste materials that have been buried for disposal (Knox et al. 2000). Plutonium, uranium, americium, neptunium, curium, and cesium are among the elements whose nuclides occur in radioactive wastes. Uranium mining activities produce large volume of residues for ore processing. These wastes contaminate adjacent land areas. They release other toxic pollutants as well. The radionuclides released from these wastes can give rise to human exposure by transport through the atmosphere, aquatic systems, or through soil subcompartments.

A catastrophic accident occurred at the Chernobyl Nuclear Power Plant in Ukraine on April 26, 1986. The explosion caused radiation to be released into the atmosphere over a 9-day period, with prevailing winds sending the plume generally in a north to northeasterly direction. The plume eventually spread over Europe, resulting in significant fallout of radiation associated with precipitation events in Austria, Switzerland, Germany, and Sweden. High levels of radioactive deposition were also reported in Italy and Britain. The releases during the accident contaminated about 125,000 km^2 of land in Belarus, Ukraine, and Russia with radiocesium levels greater than 37 $kBq\ m^{-2}$ and about 30,000 km^2 with radiostrontium greater than 10 $kBq\ m^{-2}$. Soil and water pollution from the accident is extensive, having been recorded in 22 oblasts of Russia and considerable areas of some other countries. In these areas, radionuclides have been measured in the soil to depths up to 25 cm. This quarter-meter represents the vertical zone in which crop cultivation takes place. As a result, in northern Ukraine, over 100,000 ha of agricultural land containing some of the world's richest soils has had to be abandoned (Savchenko 1995).

The movement of radionuclides in soil varies with soil type, pH, rainfall, and agricultural tilling. Radiocesium is generally confined to particles with a matrix of uranium dioxide, graphite, iron–ceramic alloys, silicate-rare earth, and silicate combinations of these materials. Cesium-137 behaves like potassium and strontium-90 behaves like calcium in soil. Cesium-137 is not adsorbed strongly in soil and is readily taken up by plants. The movement of these radionuclides in the soil depends also on the chemical breakdown of these complexes by oxidation to release more mobile forms. The bulk of the fission products are distributed between organomineral and mineral parts of the soil largely in humic complexes. Radionuclides deposition can be a significant pathway to human exposure by ingestion of contaminated pasture by animals and then by the ingestion of contaminated animal products (meat and milk). Plant foods also accumulate radionuclides.

Various soil pollutants and their sources are summarized in Table 6.4.

6.2.11 Organic Pollutants in Soil

Organic pollutants can reach the soil by dry or wet deposition after either long-range aerial transport from diffuse sources or from short-range transport from point sources such as industrial discharges, waste deposits, sludge amendments, and pesticide spraying. The contaminants are partitioned between soil particles, interstitial water, and interstitial air, and uptake by plants may occur from the water or air phases. The fate of a specific compound in a specific soil depends on the physical and chemical properties of both the compound and the soil. Organic pollutants in soil include short-chain alkanes; chlorinated solvents such as polychloroethylene (PCE, CCl_2CCl_2), trichloroethylene (TCE, C_2HCl_3), dichloroethylene (DCE, CHClCHCl), and vinyl chloride or chloroethylene (VC, CH_2CHCl); monoaromatic hydrocarbons; chlorinated

Table 6.4 Sources of pollutants in soil

Source	Materials	Pollutants
Agricultural	Phosphate fertilizers	As, Cd, Mn, U, V, and Zn
	Pig and Poultry manures	As, Cu
	Pesticides	As, Cu, Mn, Pb, Zn, organic pollutants
	Oil spillages	Hydrocarbons
Electricity generation	Fallout, Ash	Si, SO_x, NO_x, heavy metals, coal dust
Gasworks	Tars	As, Cd, Cu, Pb, $S^=$, $SO_4^=$, hydrocarbons, phenols, benzenes, xylene, naphthalene, PAHs, etc.
Metalliferous mining and smelting	Dispersed tailings, ore separates	Metals, including heavy metals, cyanides
Metallurgical industries	Metal wastes, solvents, residues, aerosols	Metals including heavy metals (As, Cd, Cr, Cu, Ni, Pb, Mn, etc.)
Chemical industries	Particulates, effluents, scraps, damaged parts	Heavy metals (As, Cd, Cr, Cu, Ni, Pb, Mn, etc.) and organic pollutants, including PAHs
Waste disposal	Sewage sludge, scrap heaps, coal ash, waste incinerators, landfills	NH_4^+, PAGHs, PCBs, NO_3^-, metals (As, Cd, Cr, Cu, Ni, Pb, Mn, etc.)
Transport	Particulates, acid deposits, solvents, oils	Pb, Br, Cd, Zn, P, As

aromatic compounds; polynuclear aromatic hydrocarbons (PAHs); and polychlorinated biphenyls (PCBs). Monoaromatic hydrocarbons include benzene, toluene, ethylbenzene, and xylenes (also collectively known as BTEX). Chlorinated aromatic compounds include hexachlorobenzene and pentachlorophenol. Among the organic inputs in soil, some are easily degraded by soil organisms. But there are many organic pollutants which are very resistant to chemical or biological degradation. They are called persistent organic pollutants (POPs). They are organic compounds that, to a varying degree, resist photolytic, biological, and chemical degradation (Pedro et al. 2006). POPs include phthalate esters (PAEs), nonylphenol ethoxylates (NPEs), PAHs, PCBs, and bisphenol A added to soils with wastewater and sewage sludge. Dioxins and furans are some toxic organic pollutants that are added to soil with herbicides and wastes from paper and wood industries. Of all the dioxins and furans, 2,3,7,8-tetrachlorodibenzo-p-dioxin (2,3,7,8 TCDD). is considered the most toxic. Crookes and Howe (1993) reported that the abundance of PCB in a highly sludge amended soil was about eight times higher than in sludge itself indicating a buildup of PCBs during many years of heavy sludge amendment at the site. PCNs (polychlorinated naphthalenes) are newly recognized persistent pollutants. Halogenated hydrocarbons, including dioxins and furans, are by far the most important group of POPs. A brief account of the major groups of persistent organic pollutants is given below.

Polycyclic aromatic hydrocarbons (PAHs) are a large group of several 100 compounds which are produced during combustion processes, with a broad range of physicochemical properties, representative of most other POPs. The International

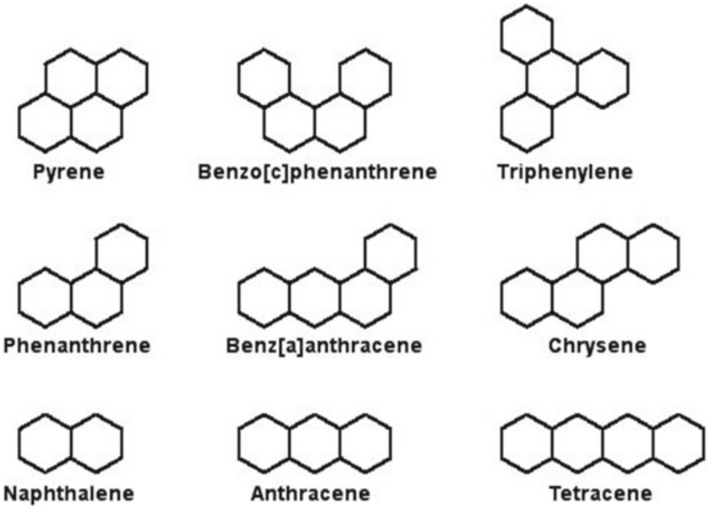


Fig. 6.15 Polycyclic aromatic hydrocarbons (PAH)

Stockholm Convention on Persistent Organic Pollutants identified 12 classes of compounds (the so-called dirty dozen) and considered as dangerous for the environment, and in 2010, the list of compounds is further expanded by nine more classes of compounds. They are all POPs, and the POPs are either intentionally produced for different purposes (e.g., many halogenated compounds for plant protection or industrial use) or unintentionally as industrial by-product or after accidents (like the dioxins and furans which are produced when halogenated compounds burn). The production of other POPs like the polycyclic aromatic hydrocarbons (PAHs) cannot even be avoided because these compounds are formed in any incomplete combustion process of organic matter (Sims and Overcash 1983). The PAHs is a ubiquitously occurring class of toxic compounds containing several hundred individual compounds. Today, anthropogenic combustion of fossil fuels is thought to be the most important source of PAH inputs to the environment. The largest environmental reservoir of PAHs is the soil (Wild and Jones 1995). Figure 6.15 shows the structures of some PAHs.

A polychlorinated biphenyl (PCB) is any of the 209 organochlorides with 2–10 chlorine atoms attached to biphenyl, which is a molecule composed of two benzene rings. Figure 6.16 shows the general structure of PCB. Due to PCBs' toxicity and classification as a persistent organic pollutant, PCB production was banned by the US Congress in 1979 and by the Stockholm Convention on Persistent Organic Pollutants in 2001 UNEP (1999). Concerns about the toxicity of PCBs are largely based on compounds within this group that share a structural similarity and toxic mode of action with dioxin. Toxic effects such as endocrine disruption and neurotoxicity are also associated with other compounds within the group.

Polychlorinated dibenzo-*p*-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) are an important group of soil contaminants, which together form a

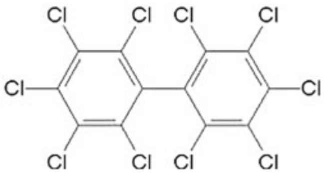
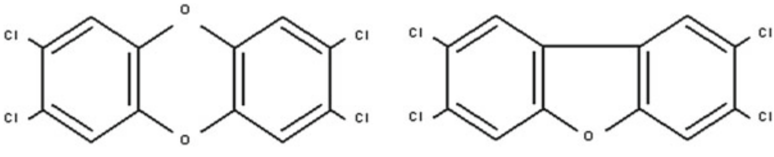
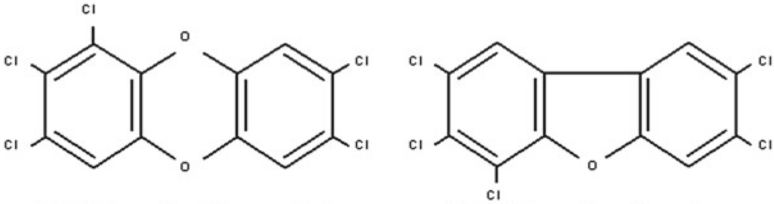


Fig. 6.16 Polychlorinated biphenyl (PCB)



2,3,7,8-Tetrachlorodibenzo-p-dioxin

2,3,7,8-Tetrachlorodibenzofuran



1,2,3,7,8-Pentachlorodibenzo-p-dioxin

2,3,4,7,8-Pentachlorodibenzofuran

Fig. 6.17 Dioxins and furans

structurally related group of 210 theoretical compounds. PCDDs and PCDFs have the general chemical formulae of $C_{12}H_8\text{-}nO_2Cl_n$ and $C_{12}H_8\text{-}nOCl_n$, respectively, with *n* representing the number of chlorine atoms (between one and eight) in the molecule (Environment Agency 2007). There are 75 different PCDD and 135 different PCDF compounds or congeners, depending on the number of chlorine atoms and ring substitution location. PCDDs and PCDFs can be divided into groups based on their degree of chlorination known as homologues; for example, all hexachlorodibenzo-p-dioxins (HxCDDs) have six chlorine atoms in the molecule. Congeners containing the same number of chlorine atoms are members of the same homologous group. Figure 6.17 shows structures of some dioxins and furans.

Of the 210 theoretical compounds, 17 PCDDs and PCDFs have been identified as being of greatest toxicological concern (HPA 2008) because of their similarity to the structure of the most toxic PCDD, which is 2,3,7,8-tetrachloro-*p*-dibenzodioxin (2,3,7,8-TCDD). A number of polychlorinated biphenyls (PCBs) are also considered to possess dioxin-like toxicity and are known as “dioxin-like PCBs” (Environment Agency 2009). PCDDs and PCDFs are not produced intentionally (except in small amounts for research purposes) but are generated as by-products of combustion and as trace contaminants during the production of organochlorine compounds such as chlorophenols and chlorobenzenes (Environment Agency 2007).

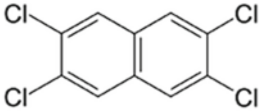


Fig. 6.18 Polychlorinated naphthalenes (PCN)

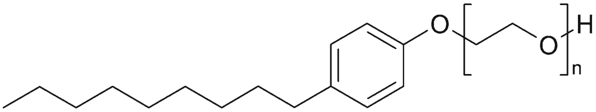


Fig. 6.19 Nonylphenol ethoxylates (NPE)

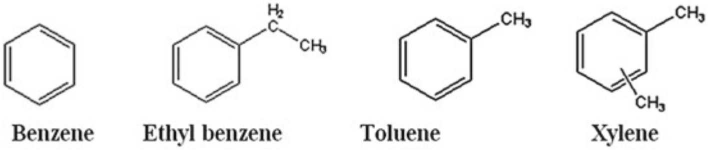


Fig. 6.20 Structures of BTEX

Chlorinated naphthalenes (CNs) and polychlorinated naphthalenes (PCNs) include 75 possible congeners and isomers. Commercial products are generally mixtures of several congeners and range from thin liquids to hard waxes to high melting point solids. Their main uses have been in cable insulation, wood preservation, engine oil additives, electroplating masking compounds, capacitors, and refractive index testing oils and as a feedstock for dye production. The major sources of release of chlorinated naphthalenes into the environment are likely to be from waste incineration and disposal of items to landfill. Chlorinated naphthalenes are expected to adsorb onto soil and sediments to a large extent. The lower chlorinated congeners are likely to show a moderate sorption tendency, and the higher chlorinated congeners are likely to show a strong sorption tendency. The structure of PCN is shown in Fig. 6.18.

Nonylphenol (NP) and nonylphenol ethoxylates (NPEs) are produced in large volumes, with uses that lead to widespread release to the terrestrial and aquatic environment. NP is persistent, bioaccumulative, and extremely toxic (PBT) to organisms. NP's main use is in the manufacture of NPEs. NPEs are nonionic surfactants that are used in a wide variety of industrial applications and consumer products. NPEs are converted to more resistant and more toxic NP in soil. NP has been detected in human breast milk, blood, and urine and is associated with reproductive and developmental effects in rodents. The structure of NPE is given in Fig. 6.19.

The BTEX group (benzene, toluene, ethylbenzene, and xylene) are naturally occurring constituents and the most hazardous components of gasoline (Fig. 6.20). Short-term (acute) hazards of lighter, more volatile and water-soluble aromatic compounds (such as benzenes, toluenes, and xylenes) include potential acute toxicity to aquatic life in the water column (especially in relatively confined areas) as well as potential inhalation hazards. Long-term (chronic) potential hazards of lighter, more volatile and water-soluble aromatic compounds include contamination of groundwater.

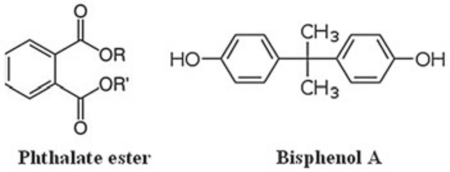


Fig. 6.21 Structures of phthalate ester and bisphenol A

Chronic effects of benzene, toluene, and xylene include changes in the liver and harmful effects on the kidneys, heart, lungs, and nervous system.

Phthalate esters (PAEs) which are widely used in plastics, automotive, clothing, cosmetics, lubricants, and pesticides industries can enter into the environment easily and regularly. PAEs (structure in Fig. 6.21) are persistent in the environment and often have cumulative effect and amplification of biological effects, which can enter the food chain and endanger human health. In general, PAEs are considered to be endocrine disrupting chemicals (EDCs), whose effects may not appear until long after exposure (Daiem et al. 2012).

6.2.11.1 Hazards of Organic Pollutants

Human health risks of organic pollutants include allergies, damage to the nervous system, disruption of the immune system, reproductive disorders, and cancer (van der Perk 2006). Some volatile organic compounds such as tetrachloroethylene, trichloroethylene, dichloroethylene, and vinyl chloride or chloroethylene, which may contaminate the soil, are potential carcinogens. Chlorinated aromatic compounds such as hexachlorobenzene and pentachlorophenol, polychlorinated biphenyls, and PAHs are also potential carcinogens. They are lipophilic and bioaccumulate in fatty human tissues. Occupational and accidental high-level exposure to some POPs is of concern for both acute and chronic worker exposure. The greatest risk is found where the use of POPs in tropical agriculture has resulted in a large number of deaths and injuries. Worker exposure to POPs during waste management is a significant source of occupational risk in many countries. Short-term exposure to high concentrations of certain POPs has been shown to result in illness and death. A study in the Philippines showed that in 1990 endosulfan became the number one cause of pesticide-related acute poisoning among subsistence rice farmers and mango sprayers. The earliest reports of exposure to persistent organic pollutants related to human health impact include an episode of HCB poisoning of food in southeast Turkey, resulting in the death of 90 % of those affected and in other exposure related incidences of hepatic cirrhosis, porphyria, and urinary, arthritic, and neurological disorders (Peters 1976). In another acute incident in Italy in 1976, release of 2,3,7,8-TCDD to the environment resulted in a purported increase of chloracne and an increased leukemia- and thyroid cancer-related mortality (Pestaori et al. 1993). More recently, the USEPA have been reviewing the dioxin-related health effects, especially for the noncarcinogenic endpoints such as immunotoxicity, reproductive diseases, and neurotoxicity.

Such examples are not common in the case of exposure to lower concentrations derived from the environment and the food chain. Studies on cell cultures in laboratory demonstrate that overexposure to certain POPs may be associated with a wide range of biological effects, including immune dysfunction, neurological deficits, reproductive anomalies, behavioral abnormalities, and carcinogenesis. Swedish investigations have reported that dietary intake of PCBs, dioxins, and furans may be linked to important reductions in the population of natural killer cells (lymphocytes) (Svensson et al. 1993), while DeWailly et al. (1993) have reported that children with high organochlorine dietary intake may experience rates of infection some 10–15 times higher than comparable children with much lower intake levels. The developing fetus and neonate are particularly vulnerable to POPs exposure, due to transplacental and lactational transfer of maternal burdens at critical periods. In another report (DeWailly et al. 1993), children in the northern Quebec region of Canada who have had significant exposure to PCBs, dioxins, and furans through breast milk also had a higher incidence of middle ear infections than children who had been bottle-fed. Studies of carcinogenesis associated with occupational exposure to 2,3,7,8, TCDD also seem to indicate that extremely high-level exposures of human populations do elevate overall cancer incidence. Some organochlorine chemicals may have carcinogenic effects and act as a strong tumor promoter. Some authors suggested a possible relationship between exposure to persistent organic pollutants and human disease and reproductive dysfunction. Sharpe and Skakkebaek observed (1993) that the increasing incidence of reproductive abnormalities in the human male might be related to increased estrogen exposure (or estrogenic type compound) and further suggest that a single maternal exposure during pregnancy of minute amounts of TCDD may increase the frequency of cryptorchidism in male offspring, with no apparent sign of intoxication in the mother. Colborn and Clement (1992) have reported that high concentrations of various persistent organic pollutants have been associated with reproductive abnormalities, including changes in the semen quality of adult rats exposed neonatally to PCBs via their mothers' milk. Associations have been suggested between human exposure to certain chlorinated organic contaminants and cancers in human populations. Preliminary evidence suggests a possible association between breast cancer and elevated concentrations of DDE, although the role of phytoestrogens and alterations in lifestyle cannot be dismissed as important risk factors in the increase in estrogen-dependent breast cancer incidence. Wolff and coworkers (1993) have reported that levels of DDE and PCBs were higher for breast cancer case patients.

6.2.11.2 Bioconcentration, Bioaccumulation, and Biomagnification of Organic Pollutants

Organic pollutants undergo a number of transformations in soils: some can be degraded biologically, photochemically, and chemically; the low molecular weight organic pollutants may volatilize, and some may be lost by leaching (Semple et al. 2003). Some organic pollutants get adsorbed onto soil minerals and organic matter (Schwarzenbach et al. 1993). A portion of organic pollutants undergoes bioaccumulation in organisms. Some examples are given below.

Organisms (flora and fauna) absorb some of the pollutants from soil and concentrate them into their tissues. The tendency of chemicals to concentrate in biota is expressed as a bioconcentration factor (BCF), defined as the ratio of the chemical concentration in biota to that in its environment at steady state (Hamelink 1977). The uptake of hydrophobic or lipophilic organic pollutants by organisms results in the accumulation of pollutants in the food chain. Plants are often used to measure BCF for soils. Bioavailability of POPs plays key role, because only bioavailable fraction of POPs can be accumulated in soil organisms. Bioconcentration is the process that causes an increased chemical concentration in an organism, say a plant, compared to that in soil, due to the uptake of chemical by absorption which can occur via the absorbing surface. In addition, pollutants enter into biota along the food chain through dietary uptake, which is referred to as biomagnification. Bioaccumulation, which is the slow buildup of organic chemicals in bodies, occurs through bioconcentration. If the concentration of a chemical in an organism is dependent on both the concentration in the medium and in the food the compound is said to bioaccumulate. Transfer of a pollutant from one trophic level to another leading to increased concentration is referred to as biomagnification. If bioaccumulation occurs in a keystone species, it can affect every other organism in its far-reaching niches. Bioaccumulation of pollutants depends on a number of factors, including biological diversity and abundance, soil characteristics including texture, pH, organic matter, CEC, and solubility, polarity, hydrophobicity, and molecular structure of the pollutant (Reid et al. 2000; Doick et al. 2005). Bioaccumulation of persistent organic pollutants (POPs) by earthworms is a very important process occurring in contaminated soils. It results to toxic effects for earthworms (while their ecological function is impaired) and to biomagnification of food chains with all related effects. Earthworms play crucial role in the soil environment contributing to the decomposition of organic matter and improving soil structure. They live in close contact with soil organic matter and are exposed to POPs accumulated there. The most important pathway for humans to bioaccumulating organic compounds in the terrestrial environment is represented by cattle through dairy products and beef (McLachlan 1996). The importance of beef and dairy products compared to fruits and vegetables can be attributed to the large quantity of herbage that must be fed to cattle to produce a given quantity of milk and beef. Organic chemicals which are lipophilic and persistent accumulate in the milk and beef fat. Grasses bioaccumulate the pollutants, and cattle biomagnify them in milk and beef. There is bioconcentration, bioaccumulation, and biomagnification along the grass–cow–human food web. Biomagnification occurs as pollutants move from a lower trophic level to a higher trophic level.

6.2.11.3 Remediation of Organic Pollutants from Soil

Physical and Chemical Methods

Based on isolation, immobilization, toxicity reduction, physical separation, extraction, and decomposition (Sheoran et al. 2008), the conventional methods of remediation of organic pollutants include soil washing/flushing, thermal desorption, chemical oxidation and decomposition, vitrification, etc. (Sharma and Reddy 2004). There are in situ (on-site) and ex situ (off-site) remediation techniques. In situ methods are used directly at the contamination site so that soil does not need to be excavated, and therefore the chance of causing further environmental harm is minimized. Most of these methods are, however, relatively expensive and slow (Ward et al. 2003) or limited by the production of secondary waste streams that require subsequent disposal or treatment.

Attempts of soil washing and solvent extraction have been made as ex situ methods. In these techniques, water and other solvent mixtures, including dichloromethane, ethanol, methanol, and toluene, have been utilized (Rababah and Matsuzawa 2002). Additionally, surfactants have also been tried. Surfactants such as TWEEN 40, TWEEN 80, Brij 30, DOWFAX 8390, and STEOL 330 have shown to be effective for PAH removal from soil (Ahn et al. 2008). Recent studies have advocated the use of vegetable oil as a nontoxic, biodegradable, and cost-effective alternative to these conventional solvents and surfactants (Gong et al. 2006). Successful removal of PAHs from soil with efficiencies above 80 % has been reported. Pizzul et al. (2007) reported the use of rapeseed oil on the degradation of polycyclic aromatic hydrocarbons in soils by *Rhodococcus wratislaviensis*.

The hydrogen peroxide oxidation technique has been employed for the remediation of organic pollutants in soils with more success. It is relatively fast, taking only days or weeks; the contaminants are treated in situ and converted to harmless substances (e.g., H_2O, CO_2, O_2, halide ions). Hydrogen peroxide can be electrochemically generated on-site, which may further increase the economic feasibility and effectiveness of this process for treatment of contaminated sites. Natural iron oxide minerals (hematite Fe_2O_3, goethite FeOOH, magnetite Fe_3O_4, and ferrihydrite) present in soil can catalyze hydrogen peroxide oxidation of organic compounds. Disadvantages include difficulties controlling in situ heat and gas production. Low soil permeability, incomplete site delineation, and soil alkalinity may limit the applicability of the hydrogen peroxide oxidation technique (Goi et al. 2009).

Permeable reactive barrier (PRB) technology, using iron metal or zerovalent iron (Fe^0) as reactive media, has been very effective in dehalogenation (detoxification) of organic contaminants in groundwater. A PRB consists of installing a trench perpendicular to the path of groundwater flow and filling it with Fe^0 (e.g., iron filings). As the contaminant-laden groundwater passes through the PRB, the organic contaminants react with Fe^0 and are dehalogenated into nontoxic forms (Gillham and O'Hannesin 1994; Sharma and Reddy 2004). Several studies investigated Fe^0 as an effective reductant in treatment of chlorinated ethylenes, halomethanes, nitroaromatic compounds, pentachlorophenol, chlorinated pesticides such as DDT, polychlorinated biphenyls, atrazine, and other organic compounds containing reducible functional groups or bonds. With this experience with Fe^0, nanotechnology has emerged as an efficient tool of remediation of soils polluted with organic pollutants. The technology involves the synthesis of nanoscale iron particles (NIP) and their application to contaminated soils. The reaction pathways of NIP with target halogenated organic contaminants are similar to that of zerovalent iron commonly used in a PRB technology. However, due to their infinitesimally small size, NIP can be highly reactive due to their high surface-to-volume ratio and greater number of reactive sites and higher intrinsic reactivity on reactive sites. In addition, NIP can be injected directly into the contaminated zones, making the in situ remediation faster and effective. Reddy (2010) has developed inexpensive and environmentally benign lactate-modified NIP that are stable and capable of transporting in soils and groundwater and dehalogenating organic pollutants such as pentachlorophenol and dinitrotoluene.

Electrokinetic (EK) remediation (Lu et al. 2005) is a green remediation technology developed recently for treatment of soils contaminated by heavy metals and organic pollutants. It has become an important development in soil remediation and has showed promising application prospects (Lageman 1993). The main advantages of EK remediation are as follows: (1) capable of using for remediation of soils with low permeability, (2) in situ remediation and decreasing on-spot pollution to the least, (3) shorting remediation time, and (4) lowering the cost. Saichek and Reddy (2005) summarized the applications of electrokinetically enhanced remediation of soils contaminated by hydrophobic organic compounds. Yap et al. (2011) reviewed Fenton-based treatments specifically for polycyclic aromatic hydrocarbon-contaminated soils. Huang et al. (2012) summarized the performances of six enhanced EK remediation methods for soils contaminated with organic pollutants: direct EK technique, EK combined with Fenton technique, EK combined with surfactants/cosolvents technique, EK combined with bioremediation method, the method of Lasagna, and EK combined with ultrasonic remediation method.

Bioremediation of Organic Pollutants

Bioremediation is defined as the elimination, attenuation, or transformation of polluting substances by the use of biological processes (Shukla et al. 2010). It uses relatively low-cost, low-technology techniques, which generally has a high public acceptance and can often be carried out on-site (Vidali 2001). It employs biological agents, mainly microorganisms, for example, yeast, fungi, or bacteria, to clean up contaminated soil. This technology includes biostimulation (stimulating viable native microbial population), bioaugmentation (artificial introduction of viable population), bioaccumulation (live cells), biosorption (dead microbial biomass), phytoremediation (using plants), and rhizoremediation (plant and microbe interaction). So, bioremediation may broadly be grouped into (1) microbial remediation, (2) phytoremediation, and (3) rhizoremediation. Here, microbial remediation will be considered as bioremediation proper. Phytoremediation and rhizoremediation will be discussed separately.

Microbial Remediation

The first patent for a biological remediation agent was registered in 1974, which was a strain of *Pseudomonas putida* (Prescott et al. 2002) able to degrade petroleum. Now, more than 100 genera of microbes are used for organic pollution control of soil. These organisms belong to at least 11 different prokaryotic divisions (Glazer and Nikaido 2007). Bioremediation has been used at a number of sites worldwide, including Europe, with varying degrees of success. Techniques are improving as greater knowledge and experience are gained, and there is no doubt that bioremediation has great potential for dealing with certain types of site contamination. A list of some contaminants potentially suitable for bioremediation is given below (Vidali 2001):

Chlorinated solvents	Trichloroethylene, perchloroethylene
Polychlorinated biphenyls (PCBs)	4-chlorobiphenyl, 4,4-dichlorobiphenyl
Chlorinated phenol	Pentachlorophenol
BTEX	Benzene, toluene, ethylbenzene, xylene
Polyaromatic hydrocarbons (PAHs)	Naphthalene, anthracene, fluorene, pyrene, benzo(a)pyrene
Pesticides	Atrazine, carbaryl, carbofuran, coumaphos, diazinon, glycophosphate, parathion, prpham,2,4D

Contaminant compounds are transformed by living organisms, individually or in mixtures, through reactions related to their metabolic processes. For bioremediation to be effective, microorganisms must enzymatically attack the pollutants and convert them to harmless products. It can be effective only where environmental conditions permit microbial growth and activity. The microorganisms employed for biodegradation of organic pollutants of soil may be classified into the following groups:

1. *Aerobic*: Efficient aerobic degraders are *Pseudomonas*, *Alcaligenes*, *Sphingomonas*, *Rhodococcus*, and *Mycobacterium*. They have been used to degrade pesticides and hydrocarbons, both alkanes and polyaromatic compounds. Many of these bacteria use the contaminant as the sole source of carbon and energy.
2. *Anaerobic*: Anaerobic bacteria are less used than aerobic bacteria. They can effectively be used for bioremediation of polychlorinated biphenyls (PCBs) in river sediments, dechlorination of the solvent trichloroethylene (TCE), and chloroform.
3. *Ligninolytic fungi*: Fungi such as the white rot fungus *Phanerochaete chrysosporium* have the ability to degrade an extremely diverse range of persistent or toxic organic pollutants. Common substrates used include straw, sawdust, or corn cobs.
4. *Methylotrophs*: Aerobic bacteria include those that utilize methane for carbon and energy. The initial enzyme in the pathway for aerobic degradation, methane mono-oxygenase, has a broad substrate range and is active against a wide range of compounds, including the chlorinated aliphatics trichloroethylene and 1,2-dichloroethane.

Degrading microbes and contaminants must be in contact for degradation to be effective. Since the microbes and the contaminants are not uniformly spread in soil, achieving an effective contact between the two is difficult. However, some bacteria are mobile and exhibit a chemotactic response, sensing the contaminant and moving toward it. Filaments of fungi grow toward the contaminant. It is possible to enhance the mobilization of the contaminant utilizing some surfactants such as sodium dodecyl sulfate (Vidali 2001).

In Situ Bioremediation

These techniques are generally the most desirable options because of lower cost and less disturbance of the soil. It does not need excavation and transport soil materials to a temporary treatment shed and spreading contaminants (USEPA 1996). It is, however, restricted by the depth of the soil. In many soils, effective oxygen diffusion for desirable rates of bioremediation extends to a range of only a few centimeters to about 30 cm into the soil. However, depths greater than 60 cm have been effectively treated in some cases. The most important in situ bioremediation methods are described in brief here.

Bioventing: This is the most common method of in situ treatment of organic contaminants. It involves the supplying of air and nutrients through wells to contaminated soil for stimulating the indigenous bacteria. Bioventing employs low air flow rates and provides only the amount of oxygen necessary for the biodegradation while minimizing volatilization and release of contaminants to the atmosphere. It works for simple hydrocarbons and can be used where the contamination is deep under the surface.

In situ biodegradation: It involves supplying oxygen and nutrients by circulating aqueous solutions through contaminated soils to stimulate naturally occurring bacteria to degrade organic contaminants. It can be used for soil and groundwater. Generally, this technique includes conditions such as the infiltration of water-containing nutrients and oxygen or other electron acceptors for groundwater treatment.

Biosparging: Biosparging involves the injection of air under pressure below the water table to increase groundwater oxygen concentrations and enhance the rate of biological degradation of contaminants by naturally occurring bacteria. Biosparging increases the mixing in the saturated zone and thereby increases the contact between soil and groundwater. The ease and low cost of installing small-diameter air injection points allows considerable flexibility in the design and construction of the system.

Bioaugmentation: Bioremediation frequently involves the addition of microorganisms indigenous or exogenous to the contaminated sites. Most soils with long-term exposure to biodegradable waste have indigenous microorganisms that are effective degraders if the land treatment unit is well managed. Some soils need the import of exogenous microorganisms, but they often suffer from competition of the native microbial community and fail to develop effective colonization.

Ex Situ Bioremediation

These techniques involve the excavation or removal of contaminated soil from ground and transporting the soil materials to a temporary treatment shed. These methods offer the risk of spreading pollutants off-site and are not suitable for large-scale remediation. The types of ex situ remediation methods are briefly discussed.

Landfarming: In this technique, contaminated soil is excavated and spread over a prepared bed and periodically tilled until pollutants are degraded. Indigenous biodegradative microorganisms are stimulated which facilitate their aerobic degradation of contaminants. The practice is limited to the treatment of superficial 10–35 cm of soil.

Composting: It involves combining contaminated soil with nonhazardous organic ammendments such as manure or agricultural wastes. The presence of these organic materials supports the development of a rich microbial population and elevated temperature characteristic of composting.

Biopiles: Biopiles are a hybrid of landfarming and composting. Essentially, engineered cells are constructed as aerated composted piles. Typically used for treatment of surface contamination with petroleum hydrocarbons, they are a refined version of landfarming that tend to control physical losses of the contaminants by leaching and volatilization. Biopiles provide a favorable environment for indigenous aerobic and anaerobic microorganisms (von Fahnestock et al. 1998).

Bioreactors: Slurry reactors or aqueous reactors are used for ex situ treatment of contaminated soil and water pumped up from a contaminated plume. It involves the processing of contaminated soil through an engineered containment system.

Phytoremediation

The use of plants for in situ treatment of pollutants in soils, sediments, and water is known as phytoremediation. Phytoremediation is a relatively new technology that offers clear advantages over traditional methods for site cleanup. Some of its applications have only been assayed at the laboratory or greenhouse level, but others have been field tested sufficiently to allow full-scale operational (Campos et al. 2008). It is best applied at sites with relatively shallow contamination of pollutants that are amenable to phytoextraction, phytovolatilization, phytodegradation, and/or phytostimulation.

Phytoextraction: the uptake of contaminants in plant roots and their concentration in harvestable tissues

Phytovolatilization: the uptake of contaminants by plants and their subsequent release into the atmosphere in a volatile form

Phytodegradation: biodegradation of pollutants by plant enzymes

Phytostimulation: biodegradation of pollutants by plants, facilitated by microorganisms in the rhizosphere

This technology is especially valuable where the contaminated soils are fragile and prone to erosion. Phytotransformation and rhizosphere bioremediation are applicable to sites contaminated with organic pollutants that include pesticides. It is cost-effective and has aesthetic advantages and long-term applicability (Brown 1995). Plants can withstand relatively high concentrations of organic chemicals without toxic effects, and they can uptake and convert chemicals quickly to less toxic metabolites in some cases. In addition, they stimulate the degradation of organic chemicals in the rhizosphere by the release of root exudates, enzymes, and the buildup of organic carbon in the soil. Different species of plants have been used in various applications including *Salix* spp. (hybrid poplars, cottonwoods, and willow), grasses (rye, Bermuda grass, sorghum, fescue, bulrush), and legumes (clover, alfalfa, and cowpeas). The main limitations of the technique are the following: (1) limited regulatory acceptance, (2) long time required for cleanup below acceptable thresholds, and (3) difficulty in establishing and maintaining vegetation at some sites.

Recent field tests of phytoremediation are reported on wastes containing petroleum hydrocarbons, such as benzene, toluene, ethylbenzene, and xylenes (BTEX), and polycyclic aromatic hydrocarbons (PAHs), pentachlorophenol, polychlorinated biphenyls (PCBs), chlorinated aliphatics (trichloroethylene, tetrachloroethylene, and 1,1,2,2-tetrachloroethane), ammunition wastes (2,4,6-trinitrotoluene or TNT, and RDX), metals (lead, cadmium, zinc, arsenic, chromium, selenium), pesticide wastes and runoff (atrazine, cyanazine, alachlor), radionuclides (cesium-137, strontium-90, and uranium), and nutrient wastes (ammonia, phosphate, and nitrate). A list of plants employed for phytoremediation of organic pollutants from soil is given below.

1. *Arabidopsis thaliana* (Campos et al. 2008)
2. *Cucurbita pepo* (Trapp and Karlson 2001)
3. *Festuca arundinacea* (Zand et al. 2010)
4. *Galega orientalis* (Trapp and Karlson 2001)
5. *Glycine max* (Njoku et al. 2009)
6. *Hamamelis virginiana* (Barnswell 2005)
7. *Ipomoea batatas* (Doty 2008)
8. *Leucaena leucocephala* (Doty et al. 2003)
9. *Morus rubra* L. (Trapp and Karlson 2001)
10. *Nicotiana tabacum* L. (Campos et al. 2008)
11. *Oryza sativa* L. (Kawahigashi et al. 2007)
12. *Populus deltoides* (Barnswell 2005)
13. *Populus* spp. (Campos et al. 2008)
14. *Quercus* spp. (Barnswell 2005)
15. *Robinia pseudoacacia* (Barnswell 2005)
16. *Salix* spp. (Campos et al. 2008)
17. *Salix viminalis* (Trapp and Karlson 2001)
18. *Senecio glaucus* (Radwan et al. 1995).
19. *Solanum tuberosum* L. (Inui et al. 2001)
20. *Ulmus pumila* (Barnswell 2005)
21. *Zea mays* (Zand et al. 2010)

Rhizoremediation

There is enhanced soil organic carbon, soil bacteria, and mycorrhizal fungi in the rhizosphere. These factors encourage degradation of organic chemicals in soil. Rhizosphere bioremediation is also a kind of plant-assisted bioremediation. Jordahl et al. (1997) showed that the numbers of beneficial bacteria increased in the root zone of hybrid poplar trees relative to an unplanted reference site. Plants may also release exudates to the soil environment that help to stimulate the degradation of organic chemicals by inducing enzyme systems of existing bacterial populations, stimulating growth of new species that are able to degrade the wastes, and/or increasing soluble substrate concentrations for all microorganisms (Barkovskii et al. 1996). Plants help with microbial transformations through the following: (1) mycorrhizal fungi and bacteria associated with plant roots metabolize the organic pollutants, (2) plant exudates stimulate bacterial transformations (enzyme induction), (3) buildup of organic carbon increases microbial mineralization rates (substrate enhancement), (4) plants provide habitat for increased microbial populations and activity, and (5) oxygen is pumped to roots ensuring aerobic transformations.

Fungi, growing in symbiotic association with the plant, have unique enzymatic pathways that help to degrade organics that could not be transformed solely by bacteria. In addition to soluble exudates, the rapid decay of fine root biomass can become an important addition of organic carbon to soils that serves to retard organic chemical transport. Microbial mineralization of atrazine is directly related to the fraction of organic carbon in the soil (McFarlane et al. 1987).

6.2.12 Heavy Metal Pollution of Soils

Although variously defined (on the basis of density, atomic number, and atomic weight), metals (or metalloids) that have a density greater than 5 g cm^{-3} and an atomic mass exceeding that of calcium are generally considered as heavy metals. The most common environmentally important heavy metals are zinc (Zn), copper (Cu), lead (Pb), cadmium (Cd), mercury (Hg), chromium (Cr), nickel (Ni), tin (Sn), silver (Ag), and the metalloid arsenic (As). Some heavy metals play an essential role in plant and animal physiology and are thus required in small amounts for normal healthy growth (Zn, Cu, and Ni for plants; Zn, Cu, Se, and Cr for animals). They are essential micronutrients. Other heavy metals are not essential and have no nutritional value (e.g., Cd, Pb, Hg, Sn, and As). All these metals are highly toxic at high concentrations or at concentrations higher than required. Some heavy metals affect the central nervous system (manganese, mercury, lead, arsenic), the kidneys or liver (mercury, lead, cadmium, copper) or skin, bones, or teeth (nickel, cadmium, copper, chromium) (Zevenhoven and Kilpinen 2001).

Soils may become polluted with heavy metals and metalloids through emissions from the rapidly expanding industrial areas, mine tailings, disposal of metal wastes, leaded gasoline and paints, application of fertilizers, animal manures, sewage sludge, pesticides, wastewater irrigation, coal combustion residues, spillage of petrochemicals, and atmospheric deposition (Khan et al. 2008; Zhang et al. 2010). Heavy metals most commonly found at contaminated sites are lead (Pb), chromium (Cr), arsenic (As), zinc (Zn), cadmium (Cd), copper (Cu), mercury (Hg), and nickel (Ni) (GWRTAC 1997). Heavy metals do not undergo microbial or chemical

Table 6.5 Some anthropogenic sources of heavy metals in soil

Sources	Metals
Landfill/chemical waste dump	As, Pb, Cr, Cd, Ba, Zn, Mn, Ni
Metal finishing/plating/electronics	Cr, Pb, Ni, Zn, Cu, Cd, Fe, As
Chemical/pharmaceutical	Pb, Cr, Cd, Hg, As, Cu
Mining/ore processing/smelting	Pb, As, Cr, Cd, Cu, Zn, Fe, Ag
Battery recycling	Pb, Cd, Ni, Cu, Zn
Wood treating	Cr, Cu, As
Nuclear processing/equipment	Ra, Th, U
Pesticides	As
Vehicle	As
Paint	Pb, Cr, Cd, Hg

degradation (Kirpichtchikova et al. 2006), and their total concentration in soils persists for a long time after their introduction (Adriano 2003). However, their chemical speciation and bioavailability may change in soil. Heavy metal contamination of soil may pose risks and hazards to humans and the ecosystem through direct ingestion or contact with contaminated soil and the food chain, drinking of contaminated water, and deterioration of food quality. Some soils are naturally metalliferous; their parent materials and rocks contain high concentrations of metals. Soils of mining areas are particularly high in heavy metals. A spectacular case was reported by Balke et al. (1973) from the area of Nievenheim in the lower reaches of the Rhine River, where a zinc processing plant has infiltrated wastewater onto the substratum. In the groundwater, the concentrations of arsenic surpassed 50 mg L^{-1}; maximum concentrations have been measured for cadmium of 600 µg L^{-1}, thallium 800 µg L^{-1}, mercury 50 µg L^{-1}, and zinc 40 mg L^{-1}. The most common anthropogenic sources of heavy metals in soil are summarized in Table 6.5.

Natural Metalliferous Soils

The most important types of natural metalliferous soils are (1) serpentine soils, developed from Fe- and Mg-rich ultramafic rocks, which are significantly enriched in Ni, Cr, and Co; (2) the calamine soils, enriched in Pb and Zn, which contain elevated Cd also and sometimes carry high concentrations of As and/or Cu; (3) Cu- and Co-containing soils such as those of the Shaban Copper Arc in the Democratic Republic of Congo, derived from argillites and dolomites containing sulfides of these metals, together with important secondary minerals (oxides, basic carbonates, silicates); and (4) soils derived from various Se-rich rocks. Serpentine soils occur over extensive areas in many countries of the world such as Japan, the Philippines, Indonesia, the USA, Turkey, Greece and the Balkan lands, parts of the European Alps, central Brazil, Cuba, Puerto Rico, and parts of Central America. Notable Se-rich soils have arisen particularly from the Cretaceous shales of the Midwestern USA (Mudgal et al. 2010).

6.2.12.1 Chemical Behavior of Heavy Metals

Soil properties that affect the behavior of heavy metals in soil include depth, slope, texture, porosity, pore size distribution, infiltration, permeability, microbial population and diversity, organic matter content, cation exchange capacity, soil pH, redox potential, and temperature. Heavy metals enter into a variety of reactions, as soil conditions permit, with the physical and chemical components of soil. These reactions affect the mobility, bioavailability, distribution, and retention of heavy metals in soil. The most common heavy metals found at contaminated sites, in order of abundance are Pb, Cr, As, Zn, Cd, Cu, and Hg (USEPA 1996). Those metals are important since they are capable of decreasing crop production due to the risk of bioaccumulation and biomagnification in the food chain. There's also the risk of superficial and groundwater contamination. Once in the soil, heavy metals are adsorbed by initial fast reactions (minutes, hours), followed by slow adsorption reactions (days, years) and are, therefore, redistributed into different chemical forms with varying bioavailability, mobility, and toxicity (Shiowatana et al. 2001; Buekers 2007). This distribution is believed to be controlled by reactions of heavy metals in soils such as (1) mineral precipitation and dissolution; (2) ion exchange, adsorption, and desorption; (3) aqueous complexation; (4) biological immobilization and mobilization; and (5) plant uptake (Levy et al. 1992).

Heavy metals in soil may be found in one or more of the following forms (Aydinalp and Marinova 2003):

(a) Dissolved (in soil solution)
(b) Exchangeable (on organic and inorganic colloidal surfaces)
(c) As structural components of the lattices of soil minerals
(d) As insoluble precipitates with other soil components as in other metal oxides and carbonates

The first two forms are mobile or labile, while the other two are currently unavailable and potentially available in the longer term. The easily soluble fractions are of the greatest interest, as their highest mobility, bioavailability, or toxicity can influence the quality of environment. Extractants including distilled water and several salt solutions (KCl, $CaCl_2$, NH_4OAc, NH_4NO_3, EDTA, and DTPA) have been used to extract the mobile or bioavailable forms of heavy metals with single extraction procedures (Karczewska et al. 1998). The mobility of metals is strongly affected by soil properties and may vary significantly in relation to changes in soil pH and organic carbon content. Other soil properties, such as cation exchange capacity (CEC), organic matter content, quantity and type of clay minerals, the content of the oxides of iron (Fe), aluminum (Al), and manganese (Mn), and the redox potential determine the soil's ability to retain and immobilize heavy metals.

Mobility of heavy metals in soil depends greatly on soil pH. Fuller (1977) considered that in acidic soils (pH 4.2–6.6) the elements Cd, Ni, and Zn are highly mobile, Cr is moderately mobile, and Cu and Pb are practically immobile, and in neutral to alkaline soils (pH 6.7–7.8), Cr is highly mobile, Cd and Zn are moderately mobile, and Ni is immobile. Humic substances form metal–organic complexes which influence metal mobilization in soils. Using gel permeation chromatography, the important role of organic matter in solubilizing heavy metals in sewage sludges and in agricultural soils amended with liquid manure can be clearly shown. Aluminum mobilization in Spodosols was demonstrated to be largely determined by the formation, solubilization, migration, and subsequent precipitation of organic complexes. Organic matter plays an important role not in forming complexes and also in retaining heavy metals in an exchangeable form. Copper is bound and rendered unavailable chiefly through the formation of complexes, while Cd is retained in an exchangeable form and is more readily available. When mobility increases, the quantities of heavy metals available to plants increase, resulting in the appearance of toxicity phenomena.

Arsenic

Arsenic is an element in group VA and period IV of the periodic table. It is a metalloid having atomic number 33, atomic mass 75, density 5.72 g cm^{-3}, melting point 817 °C, and boiling point 613 °C and can be present in several oxidation states (–III, 0, III, V) (Smith et al. 1995). It naturally occurs in a wide variety of minerals, mainly as arsenolite As_2O_3. It occurs in ores containing Cu, Pb, Zn, Ag, and Au. In aerobic environments, As (V) is dominant, usually in the form of arsenate (AsO_4^{3-}) in various protonation states: H_3AsO_4, $H_2AsO_4^-$, $HAsO_4^{2-}$, and AsO_4^{3-}. Arsenate and other anionic forms of arsenic behave as chelates and can precipitate when metal cations are present (Bodek et al. 1988). Arsenic (V) is adsorbed on iron oxyhydroxides and can coprecipitate with them under acidic and moderately reducing conditions. Coprecipitates are immobile under these conditions, but arsenic mobility increases as pH increases (Smith et al. 1995). Under reducing conditions, As(III) dominates, existing as arsenite (AsO_3^{3-}) and its protonated forms H_3AsO_3, $H_2AsO_3^-$, and $HAsO_3^{2-}$. Arsenite has a high affinity for sulfur compounds and can adsorb or coprecipitate with metal sulfides. Elemental arsenic and arsine, AsH_3, may be present under extreme reducing conditions. Methylation of arsenic forms methylated derivatives of arsine, including highly volatile dimethyl arsine $HAs(CH_3)_2$ and trimethylarsine $As(CH_3)_3$. Arsenic may be present in organometallic forms such as methylarsinic acid $(CH_3)AsO_2H_2$ and dimethylarsinic acid $(CH_3)_2AsO_2H$.

All rocks contain some arsenic, typically 1–5 mg kg^{-1}. Some igneous and sedimentary rocks may contain higher As concentrations. Principal arsenic bearing minerals are arsenopyrite (AsFeS), realgar (AsS), and orpiment (As_2S_3). The sources of As in soils are both geochemical and anthropogenic. Metallic arsenic is mainly used for strengthening alloys of copper and lead. Arsenic is used in semiconductor electronic devices. Arsenic compounds are used for production of pesticides, herbicides, and insecticides. Several organoarsenic compounds were developed as chemical warfare agents during World War I, including vesicants such as lewisite and vomiting agents such as adamsite. Residues of these materials have found their way to the soil (Smith et al. 1998a, b). Arsenic is a notorious pollutant of groundwater which is used for irrigation of extensive rice fields in Southeast Asia.

Soils may contain 1–40 mg As kg^{-1}, with an average of 5–6 mg kg^{-1}. Indiscriminate use of arsenical pesticides during the early to mid-1900s has led to extensive contamination of soils worldwide. Contamination in excess of 1,000 mg As kg^{-1} has been recorded at many sites throughout Australia. Similar contaminated sites also exist in the USA, Africa, and other parts of the world (Smith et al. 1998a, b). The main sources of As in soils is the parent materials from which the soil is derived (Yan-Chu 1994). The distribution of As in soils may vary with soil type, depending on the nature of the parent material. Background concentrations do not generally exceed 15 mg As kg^{-1} (NRCC 1978), although concentrations ranging from 0.2 to 40 mg As kg^{-1} soil have been reported (Walsh et al. 1977). Dudas and Pawluk (1980) reported background As concentrations that averaged 5 mg As kg^{-1} in 78 soil samples in Alberta. Much higher As concentrations have been reported in acid sulfate soils developed on pyritic parent material. For instance, Dudas (1987) attributed elevated As concentrations that ranged from 8 to 40 mg As kg^{-1} in Canadian acid sulfate soils to the weathering of pyrites in the parent material. Many different As compounds have been identified in the soil environment, and they may be classified into two major groups: inorganic As compounds and organic As compounds. As (V) and As (III) are the most important inorganic As species in the soil, because their compounds are highly soluble in water (Masscheleyn et al. 1991).

Arsenic Contamination to Groundwater and Arsenicosis in Asia

Arsenic contamination in groundwater has invaded in several regions of the Southeast Asian countries like Bangladesh, China, India, and Nepal. Water drawn from shallow aquifers through shallow tube wells exceeds the national standards (0.050 mg L^{-1}) and USEPA and WHO standards (0.01 mg L^{-1}) of As level for drinking water. The cause of this contamination to groundwater is geochemical; the sedimentary rocks there contain high As. People use shallow tube well water for domestic uses including drinking. Large-scale occurrences of arsenicosis have been reported from the contaminated areas. Health problems associated with elevated levels of arsenic include thickening and discoloration of the skin, stomach pain, nausea, vomiting, diarrhea, numbness in hands and feet, partial paralysis, and blindness. Arsenic has been linked to cancer of the bladder, lungs, skin, kidney, nasal passages, liver, and prostate. Thirty million people in Bangladesh use arsenic contaminated water for drinking. About 10,000–30,000 people have been diagnosed with arsenicosis (Alam et al. 2002). In China, endemic arsenicosis was found successively in many areas in the mainland during 1980s, and more than 10,000 arsenicosis patients were diagnosed by 2001. The main As-contaminated provinces are Xinjiang, Inner Mongolia, Shanxi, Ningxia, Jilin, and Qinghai. In India, arsenic-contaminated groundwater occurs mainly in the states of West Bengal, Bihar, Uttar Pradesh, Assam, Jharkhand, Chhattisgarh, and Madhya Pradesh. According to UNICEF, over 13.8 million people are at risk of arsenicosis in West Bengal (Chakraborti et al. 2002). The problems are mainly related with As-contaminated drinking water, but there is also the risk of contaminating the food chain. Shallow aquifer water is the major source of irrigation in rice growing areas of Asia.

Cadmium

Cadmium (Cd) is a lustrous, silver-white, ductile, very malleable, and soft metal. Cadmium is a metallic element with atomic number 48, atomic weight 112.4, density 8.65 g cm^{-3}, melting point 320.9 °C, and boiling point 765 °C. Cadmium has a relatively low crustal abundance, although it occurs ubiquitously in rocks and soils (Alloway 1995). It is rarely found in its elemental form. The main minerals containing cadmium are greenockite (CdS), octavite (CdSe), and monteponite (CdO) (Kabata-Pendias and Mukherjee 2007). Cadmium is often found in association with zinc ores and zinc-bearing lead ores. Its compounds almost exclusively involve the +2 oxidation state. Cadmium forms simple salts with oxygen, sulfur, and many common anions such as chloride, nitrate, and carbonate. Cadmium often forms simple hydrated hydroxyl ions such as $[Cd(OH)(H_2O)x]^+$ in aqueous solution. It also has appreciable coordination chemistry with ligands including halides, hydroxides, cyanides, and nitrate (Alloway 1995). Organocadmium compounds are rather reactive and unstable.

Cadmium is regarded as one of the most toxic trace elements in the environment. The increased emissions with its persistence and its relatively rapid uptake and accumulation by food chain crops contribute to its potential environmental hazards. According to Heinrichs et al. (1980), the average concentration of cadmium in the lithosphere is 0.098 mg kg^{-1}. Most mafic and granitic rocks contain a very low concentration of cadmium (<0.1 mg kg^{-1}). It has relatively high concentrations (about 10 mg kg^{-1}) in sedimentary rocks. Some shales are unusually high in Cd (>100 mg kg^{-1}).

Amounts of indigenous cadmium in cultivated and noncultivated soils are determined by the quantities of Cd in the parent materials together with amounts added through atmospheric deposition, fertilizers, pesticides, and irrigation water. Generally soils of similar age derived from igneous rocks contain the lowest Cd, soils derived from metamorphic rocks have intermediate, and soils derived from sedimentary rocks contain the largest quantities of Cd. Non-contaminated soils contain very low concentration of Cd. In a survey, a total of 3,305 soil samples were collected from crop-producing areas in 36 states of USA. Concentrations of Cd ranged from 0.005 to 2.40 mg kg^{-1}, with mean and median values of 0.27 and 0.20 mg kg^{-1}, respectively (Holmgren et al. 1986). Surface soils around mining areas contain high concentrations of Cd. For example, Munshower (1977) reported 29 mg Cd kg^{-1} in surface soil at 2 km northeast of a lead–zinc smelter in Montana. The Cd concentration of surface soil decreased with distance and reached near background levels 24 km from the source. Buchauer (1973) reported concentrations of Cd in the surface organic horizon as great as 750 mg kg^{-1}. The soils contaminated by mining operations will remain high in Cd more or less permanently because Cd is quite immobile in soil. Cadmium concentration in soil solution is relatively low ranging from 0.2 to 6.0 μg L^{-1} (Kabata-Pendias and Pendias 2000).

The Itai-Itai Episode in Japan

Cadmium concentration in food chain crops grown in non-contaminated soils is generally low (0.01–1.0 mg kg^{-1}). Leafy vegetables (lettuce, spinach) have the highest; grains (wheat, oats, barley) show the lowest, and root vegetables (carrot, radish, onion, potato) have intermediate concentrations of cadmium in edible parts. Concentrations of Cd in crops grown on Cd-contaminated soils may accumulate substantially greater than 1.0 mg kg^{-1} Cd. Consumption of cadmium-contaminated food grains may cause severe Cd poisoning.

The well-known episode of Cd poisoning of humans, the itai-itai disease, was reported in Japan in the mid-1950s (Tsuchiya 1978). The name of the disease came from the screams due to the severe pains in the bones caused by cadmium poisoning. It first impairs kidney function and progressively causes osteomalacia. The outbreak of itai-itai started around 1912 and continued until 1945. The disease was thought to be nutritional and pathological at first. After many years of investigation, the Ministry of Health and Welfare officially announced in 1968 that the disease was due to chronic Cd poisoning, and the source of cadmium was the mines in the mountainous areas of Toyama Prefecture.

Mining operations were being done for centuries in the Toyama Prefecture of Japan. In early 1900s, the Kamioka Mining Co. Ltd. was extracting zinc ores in these mines for a long time. Wastewater of these operations was released in the Jinzū River, the water of which became seriously contaminated with cadmium. This water was used by local community people for their domestic need and irrigation in paddy fields. Soils and rice grains, grown in these paddies, were also contaminated with cadmium. Since the per capita dietary consumption of rice among the Japanese population is high, the impact of Cd enrichment in the rice on the indigenous population was proportionally magnified.

As with all cationic metals, the chemistry of cadmium in the soil environment is to a great extent controlled by pH. Under acidic conditions, Cd solubility increases and very little adsorption of Cd by soil colloids, hydrous oxides, and organic matter takes place. At pH values greater than 6 units, Cd is adsorbed by the soil solid phase or is precipitated, and the concentrations of dissolved Cd are greatly reduced. Cadmium forms soluble complexes with inorganic and organic ligands, in particular with chloride ions. The formation of these complexes will increase Cd mobility in soils. Chloride can be expected to form a soluble complex with Cd^{2+} as $CdCl^{+}$, thereby decreasing the adsorption of Cd^{2+} to soil particles. In contrast to inorganic ligand ions, Cd^{2+} adsorption by kaolinite could be enhanced by the presence of organic matter via the formation of an adsorbed organic layer on the clay surface (Adriano et al. 2005). Cadmium mobility and bioavailability are higher in noncalcareous than in calcareous soils. Liming of soil raises the pH, increasing cadmium adsorption to the soil and reducing bioavailability. A general trend emerges that toxicity increases in soil when mobility of cadmium is higher; that is, soil toxicity increases as soil pH, or soil organic matter, decreases.

Cadmium may be adsorbed by clay minerals, carbonates, or hydrous oxides of iron and manganese or may be precipitated as cadmium carbonate, hydroxide, and phosphate. Adsorption mechanisms may be the primary source of cadmium removal from soils. In soils and sediments polluted with metal wastes, the greatest percentage of total cadmium was associated with the exchangeable fraction. Cadmium concentrations have been shown to be limited by cadmium carbonate in neutral and alkaline soils.

Cadmium in the human body is known to affect several enzymes. It is believed that the renal damage that results in proteinuria is the result of Cd adversely affecting enzymes responsible for reabsorption of proteins in kidney tubules. Cadmium also reduces the activity of delta-aminolevulinic acid synthetase, arylsulfatase, alcohol dehydrogenase, and lipoamide dehydrogenase, whereas it enhances the activity of delta-aminolevulinic acid dehydratase, pyruvate dehydrogenase, and pyruvate decarboxylase (Manahan 2003). The most spectacular and publicized occurrence of cadmium poisoning resulted from dietary intake of cadmium by people in the Jinzū River Valley, near Fuchu, Japan.

Lead

Lead (Pb) is a bright and silvery metal which is soft, ductile, malleable, and resistant to corrosion. Lead belongs to group IV and period 6 of the periodic table. It has atomic number 82, atomic mass 207.2, density 11.4 g cm^{-3}, melting point 327.4 °C, and boiling point 1,725 °C. Lead is generally found as a mineral combined with other elements, such as sulfur (i.e., PbS, $PbSO_4$) or oxygen ($PbCO_3$). It ranges from 10 to 30 mg kg^{-1} in the earth's crust (USDHHS 1999). Typical mean Pb concentration for surface soils worldwide averages 32 mg kg^{-1} and ranges from 10 to 67 mg kg^{-1} (Kabata-Pendias and Pendias 2001). Some metals are alloyed with lead; they are antimony (in storage batteries), calcium (Ca) and tin (Sn) (in maintenance-free storage batteries), silver (Ag) (for solder and anodes), strontium (Sr) and Sn (as anodes in electrowinning processes), tellurium (Te) (pipe and sheet in chemical installations and nuclear shielding), Sn (solders), and antimony (Sb) and Sn (sleeve bearings, printing, and high-detail castings) (Manahan 2003). The general forms of lead that are released to the environment are ionic lead, Pb(II), lead oxides and hydroxides, and lead–metal oxyanion complexes. The most stable forms of lead are Pb(II) and lead–hydroxy complexes. The predominant insoluble Pb compounds are lead phosphates, lead carbonates, and lead hydroxides. Lead sulfide (PbS) which is the most stable solid form in soil forms under reduced conditions. Under anaerobic conditions a volatile organolead (tetramethyl lead) can be formed due to microbial alkylation. Lead (II) compounds are predominantly ionic (e.g., Pb^{2+} SO_4^{2-}), whereas Pb(IV) compounds tend to be covalent (e.g., tetraethyl lead, $Pb(C_2H_5)_4$). Some Pb (IV) compounds, such as PbO_2, are strong oxidants. In addition to the inorganic compounds of lead, there are a number of organolead compounds such as tetraethyl lead which was earlier widely used as a gasoline additive.

More than 1,000 organolead compounds have been synthesized for commercial and toxicological reasons. They are largely limited to the alkyl (methyl and ethyl) lead compounds and their salts. Inhalation and ingestion are the two routes of exposure, and the effects from both are the same. Pb accumulates in the body organs (i.e., brain), which may lead to poisoning (plumbism) or even death. The gastrointestinal tract, kidneys, and central nervous system are also affected by the presence of lead. Children exposed to lead are at risk for impaired development, lower IQ, shortened attention span, hyperactivity, and mental deterioration. Adults usually experience decreased reaction time, loss of memory, nausea, insomnia, anorexia, and weakness of the joints when exposed to lead. Lead can cause serious injury to the brain, nervous system, red blood cells, and kidneys (Baldwin and Marshall 1999). Lead performs no known essential function in the human body, it can merely do harm after uptake from food, air, or water. The most serious source of exposure to soil lead is through direct ingestion (eating) of contaminated soil or dust. In general, plants do not absorb or accumulate lead. However, in soils testing high in lead, it is possible for some lead to be taken up. Studies have shown that lead does not readily accumulate in the fruiting parts of vegetable and fruit crops (e.g., corn, beans, squash, tomatoes, strawberries, and apples). Higher concentrations are more likely to be found in leafy vegetables (e.g., lettuce) and on the surface of root crops (e.g., carrots). Since plants do not take up large quantities of soil lead, the lead levels in soil considered safe for plants will be much higher than soil lead levels where eating of soil is a concern (pica). Generally, it has been considered safe to use garden produce grown in soils with total lead levels less than 300 ppm. The risk of lead poisoning through the food chain increases as the soil lead level rises above this concentration. Even at soil levels above 300 ppm, most of the risk is from lead-contaminated soil or dust deposits on the plants rather than from uptake of lead by the plant (Rosen 2002).

The proportion of total soil Pb that exists in the soil solution is very small. Alloway et al. (1985) found Pb concentrations ranging from 61 to 12,537 μg L^{-1} in soil solutions enriched from a weathering Pb ore and soil contaminated by Pb/Zn mining and sewage sludge. Lead may exist in the soil solution as the free metal ion, Pb^{2+}, and soluble organic and inorganic complexes. The speciation in the soil solution depends to a large extent on pH, dissolved organic carbon, and the presence of ligands. Lead activity decreases with increasing soil pH. It is widely acknowledged that Pb^{2+} is the most significant Pb species in solution at pH values less than either 7 (Wang and Benoit 1997) or 8 (Lindsay 1979). The dominant form of Pb in solution at pH values greater than 7 is $PbCO_3$ (Wang and Benoit 1997). Other minor Pb species include $PbOH^+$, $Pb(OH)_2$, $Pb(OH)_3$, $PbCl_3$, $PbCl^+$, $PbNO_3^+$, and $Pb(CO_3)_2$ (Wang and Benoit 1997). Lead exists as Pb^{2+} in solution at pH 4 (Markus and McBratney 2001).

Mercury

Mercury belongs to the same group of the periodic table with Zn and Cd. It is the only liquid metal at standard temperature and pressure. It has atomic number 80, atomic weight 200.6, density 13.6 g cm^{-3}, melting point 13.6 °C, and boiling point 357 °C. It is generally obtained as a by-product of ore processing (Smith et al. 1995). After its release to the environment, Hg usually exists in mercuric (Hg^{2+}), mercurous (Hg_2^{2+}), elemental (Hg^0), or alkylated form (methyl/ethyl mercury). The redox potential and pH of the system determine the stable forms of Hg that will be present. Mercurous and mercuric mercury are more stable under oxidizing conditions. When mildly reducing conditions exist, organic or inorganic Hg may be reduced to elemental Hg, which may then be converted to alkylated forms by biotic or abiotic processes. Mercury is most toxic in its alkylated forms which are soluble in water and volatile in air (Smith et al. 1995).

Almost all commercial production of mercury comes from the sulfide ore, cinnabar, which results from hydrothermal mineralization associated with volcanic activity. Mercury abundance in the earth's crust is very low being in the range of 0.02–0.06 mg kg^{-1}, although it is likely to be more concentrated in argillaceous sediments and in coal (Kabata-Pendias and Mukherjee 2007). The total mercury concentrations in soils of UK were reported to range from 0.07 to 1.22 mg kg^{-1} with a mean value of 0.13 mg kg^{-1}. Urban soils were found to contain higher total mercury concentrations ranging from 0.07 to 1.53 mg kg^{-1}, with a mean of 0.35 mg kg^{-1} (Environment Agency 2007).

Mercury readily forms amalgams with sodium and zinc. Mercury forms inorganic compounds in both the Hg (I) and Hg (II) valent states. A large number of organomercury compounds are also known and can be synthesized by the action of sodium amalgam or Grignard reagent and $HgCl_2$ with halogenated hydrocarbons (Greenwood and Earnshaw 1997). Mercury is most commonly encountered in the environment in elemental form, as inorganic mercuric compounds or as monomethylmercury compounds with the general formula, CH_3HgX. The most important source of mercury is the naturally occurring mineral cinnabar (HgS). Monomethylated mercury compounds are most likely to be found in soil as a result of natural microbial transformation of inorganic mercury.

Elemental Hg is stable in the soil environment. It has a strong tendency to form complexes with other anions (such as Cl^-, OH^-, and S^{2-}) and humic matter. The mercuric cation (Hg^{2+}) is rarely found in soil solution under natural conditions and the major fraction is bound in soil minerals or adsorbed either to inorganic mineral surfaces or to organic matter. Yin et al. (1996) observed in experimental studies that soil adsorption decreased significantly above pH 5 as a result of increasing amounts of dissolved organic matter and the tendency for mercury to complex strongly to organic carbon.

Inorganic mercury can be methylated by abiotic and microbial processes in soil systems and is the primary source of methylmercury compounds in soil (ATSDR 1999). In surface soils, about 1–3 % of total mercury is in the methylated form with the rest predominantly as mercuric compounds (Kabata-Pendias and Mukherjee 2007). Dimethylmercury is a highly toxic and volatile compound and is readily lost from soil to air. Monomethylated mercury compounds (CH_3HgX) are also volatile, and due to their relatively high mobility compared with inorganic forms, they are the most important mercury species for environmental pollution (ATSDR 1999).

Chromium

Chromium is a transition metal of group VIB in the periodic table with the following properties: atomic number 24, atomic mass 52, density 7.19 g cm^{-3}, melting point 1,875 °C, and boiling point 2,665 °C. It is one of the less common elements and does not occur naturally in elemental form. Chromium is mined as a primary ore product in the form of the mineral chromite, $FeCr_2O_4$. Major sources of Cr-contamination include releases from electroplating processes and the disposal of Cr containing wastes (Smith et al. 1995). Chromium (VI) is the form of Cr commonly found at contaminated sites. Chromium can also occur in the + III oxidation state, depending on pH and redox conditions. Chromium (VI) is the dominant form of Cr in shallow aquifers where aerobic conditions exist. Chromium (VI) can be reduced to Cr (III) by soil organic matter, S^{2-} and Fe^{2+} ions under anaerobic conditions often encountered in deeper groundwater. Major Cr (VI) species include chromate (CrO_4^{2-}) and dichromate ($Cr_2O_7^{2-}$) which precipitate readily in the presence of metal cations (especially Ba^{2+}, Pb^{2+}, and Ag^+). Chromate and dichromate also adsorb on soil surfaces, especially iron and aluminum oxides. Chromium (III) is the dominant form of Cr at low pH (<4). Cr^{3+} forms solution complexes with NH_3, OH^-, Cl^-, F^-, CN^-, SO_4^{2-}, and soluble organic ligands. Chromium (VI) is the more toxic form of chromium and is also more mobile. Chromium (III) mobility is decreased by adsorption to clays and oxide minerals below pH 5 and low solubility above pH 5 due to the formation of $Cr(OH)_3(s)$ (Chrostowski et al. 1991). Chromium mobility depends on sorption characteristics of the soil, including clay content, iron oxide content, and the amount of organic matter present. Chromium can be transported by surface runoff to surface waters in its soluble or precipitated form. Soluble and un-adsorbed chromium complexes can leach from soil into groundwater. The leachability of Cr (VI) increases as soil pH increases.

Nickel

Nickel has atomic number 28 and atomic weight 58.69. In low pH regions, the metal exists in the form of the nickelous ion, Ni(II). It precipitates as nickelous hydroxide, $Ni(OH)_2$, in neutral to slightly alkaline solutions. The precipitate dissolves in acid solutions, forming Ni (III), and in very alkaline conditions, it forms nickelite ion, $HNiO_2$, that is soluble in water. In very oxidizing and alkaline conditions, nickel exists in form of the stable nickelo-nickelic oxide, Ni_3O_4, that is soluble in acid solutions. Other nickel oxides such as nickelic oxide, Ni_2O_3, and nickel peroxide, NiO_2, are unstable in alkaline solutions and decompose by giving off oxygen. In acidic regions, however, these solids dissolve, producing Ni^{2+}. Nickel occurs in the environment only at very low levels and is essential in small doses, but it can be dangerous when the maximum tolerable amounts are exceeded (Wuana and Okieimen 2011). This can cause various kinds of cancer on different sites within the bodies of animals, mainly of those that live near refineries. The major sources of nickel contamination in the soil are metal plating industries, combustion of fossil fuels, and nickel mining and electroplating. In acidic soils, however, Ni becomes more mobile and often leaches down to the adjacent groundwater. Microorganisms can also suffer from Ni toxicity.

Table 6.6 Typical background and unsafe soil levels of heavy metals

Heavy metal	Typical background levels for non-contaminated soil (mg kg^{-1})	Unsafe for leafy or root vegetables (mg kg^{-1})	Unsafe for gardens and children contact (mg kg^{-1})
Arsenic	3–12	>50	>200
Cadmium	0.1–1.0	>10	>50
Copper	1–50	>200	>500
Lead	10–70	>500	>1,000
Nickel	0.5–50	>00	>500
Selenium	0.1–3.9	>50	>200
Zinc	9–125	>200	>500

Source: http://www.aleastern.com/forms/Heavy%20Metal%20Interpretation.pdf

Table 6.7 Threshold levels of heavy metals in soil

Heavy metals	Threshold levels in soils of concern for human health (mg kg^{-1})	Heavy metals	Threshold levels in soils of concern for human health (mg kg^{-1})
Arsenic	0.07	Molybdenum	380
Cadmium	1.70	Nickel	1,600
Chromium	1,00,000	Selenium	380
Cobalt	660	Silver	380
Copper	3,000	Vanadium	530
Lead	80	Zinc	23,000
Mercury	18		

Source: California Office of Human Health Assessment, http//:oehha.ca.gov/risk/chhsltable.html

6.2.12.2 Safe Limits of Heavy Metals in Soil

The term “safe level” needs to be explained. A level safe for growing vegetables may not be safe for children’s contact. A level safe for human may be toxic for crops. Moreover, getting a universally accepted safe limit of heavy metals in soil is difficult. Table 6.6 gives some typical background levels in non-contaminated soils of some common heavy metals and their unsafe levels for vegetable growing and gardening.

Zhou et al. (2008) proposed that “maximum allowable concentrations” of Cd, Cu, and Hg in Chinese agricultural soils are 0.30, 50.00, and 0.30 mg kg^{-1}, respectively. According to Coskun et al. (2006), “world limits” of Cd, Hg, Pb, and As in soil are 0.35, 0.30, 35.00, and 10.00 mg kg^{-1}, respectively. The toxic limits in soils of As, Cd, Cr, Cu, Pb, Hg, Mo, Ni, Se, and Zn are 1–4, 5–20, 5–30, 20–100, 5–30, 1–3, 10–100, 30–300, 5–10, and 100–400 mg kg^{-1}, respectively (Alloway 1995). Threshold levels of heavy metals in soils of concern for human health are given in Table 6.7.

6.2.12.3 Uptake of Heavy Metals by Plants

Plants absorb heavy metals through roots from soil and by leaves from air. Once absorbed, there are a variety of mechanisms to distribute metals within the plant body. A portion of absorbed metals may be retained in roots, barred by membranes, precipitated and accumulated in vacuoles, or translocated to shoots where they perform physiological functions if they are nutrients or beneficial elements or interfere with normal physiological functions leading to toxicities. Some plants prevent metal toxicity, by metal compartmentalization and binding to intracellular ligands. In addition to the organic acids, plants have phytochelatins and metallothioneins. Phytochelatins are a family of peptides, while metallothioneins are similar to phytochelatins in being Cys-rich, metal-complexing ligands.

According to Cho et al.(2003), the sensitivity of plants to heavy metals depends on the (1) uptake and accumulation of metals through binding to extracellular exudates and cell wall constituents; (2) efflux of heavy metals from cytoplasm to extranuclear compartments, including vacuoles; (3) complexation of heavy metal ions inside the cell by various substances, for example, organic acids, amino acids, phytochelatins, and metallothioneins; (4) accumulation of osmolytes and osmoprotectants and induction of antioxidative enzymes; and (5) activation or modification of plant metabolism to allow adequate functioning of metabolic pathways and rapid repair of damaged cell structures. Yet, metals are accumulated in plants and become toxic; and for these reasons, the yield of crop is reduced and the crop is rendered unsafe if grown in a contaminated soil.

Arsenic Uptake

Gulz (2002) performed a series of pot culture greenhouse experiments in connection with her Ph.D. work in Dipl.-Geogr., University of Munich on the uptake of As by maize, ryegrass, rape, and sunflower from arsenic-contaminated soils. Most As was accumulated in the roots of plants. Although accumulation in aboveground biomass remained much lower, As concentrations in stems, leaves, and seeds reached values above the Swiss tolerance limits for food or fodder crops (0.2 and 4 mg As kg^{-1}, respectively), except for maize. Results suggested that besides As solubility, phosphorous availability as well as phosphorous demand of the plants has to be taken into account to predict As uptake of crops. Due to the high As translocation of sunflower from roots to shoots, further experiments were performed by using this plant species. In batch as well as growth chamber experiments, the effects of phosphorous (P) fertilization on P and As availability in the soil, As uptake by sunflower, and biomass production were investigated. Already the addition of a base P fertilization (56 mg P kg^{-1}) led to a significant increase of soluble P and As concentrations in the soil. P addition further tended to increase As uptake in the roots and shoots of sunflower at low soluble As concentrations. The P effect on soluble As was lower in the silty sand than in the sandy loam, indicating that a high fixation capacity of the soil lessened the mobilizing effect of P.

Cadmium Uptake

Plant roots readily absorb cadmium from soil if it is available and translocate it to the aboveground parts. Cadmium concentrations are typically higher in the plant leaves than in fruits or storage organs, indicating that cadmium is highly mobile in plant tissues. The uptake of Cd increases proportionally to increasing soil Cd levels. A linear increase in uptake of Cd has been observed in different greenhouse studies and field trials (Brown et al. 1998; Kadar 1995). The Cd transfer factor (TF) is the ratio of Cd concentration in the plant to that in the soil. Generally, leafy vegetables have higher Cd concentrations and TF than storage organs or fruits. The TF concept suggests that plant Cd can be properly predicted from soil Cd. However, the TF varies with soil properties. Field surveys carried out in different parts of the world showed that soil Cd usually explains less than 20 % of the variability of crop Cd (Eriksson et al. 1996). It is often observed that Cd concentration in soil solution or Cd concentrations in neutral salt extracts of soil (NH_4NO_3, $NaNO_3$, or $CaCl_2$ extracts) are better predictors for crop Cd than total soil Cd (McLaughlin et al. 2000). This indicates that Cd availability is linked with Cd mobility. However, mobility and plant availability do not always go hand in hand. Soil pH has a negative relationship with Cd uptake. On the other hand, soil salinity and lime-induced zinc deficiency enhance uptake of Cd by plants (Smolders 2001). Some agricultural crops usually contain high Cd concentrations. Durum wheat, sunflower kernels, and flax have been identified as high Cd crops compared to spring wheat, barley, corn, or oats (Li et al. 1994).

Lead Uptake

There are various complexes of lead in soil, and only a small fraction of the lead present in these complexes in the soil solution are phyto-available. It has no essential function in plants; still, lead is absorbed by them mainly through the roots from soil solution and enters the food chain. Roots absorb lead via the apoplastic pathway or via Ca^{2+} permeable channels. The factors that affect lead availability and lead uptake by plants are speciation of lead, soil pH, soil particle size, cation exchange capacity, root surface area, root exudation, and degree of mycorrhizal transpiration. After uptake, lead primarily accumulates in root cells, because of the blockage by Casparian strips within the endodermis. Lead is also trapped by the negative charges that exist on roots cell walls. Excessive lead accumulation in plant tissue impairs various morphological, physiological, and biochemical functions in plants, either directly or indirectly, and induces a range of deleterious effects. It causes phytotoxicity by changing cell membrane permeability, by reacting with active groups of different enzymes involved in plant metabolism and by reacting with the phosphate groups of ADP or ATP, and by replacing essential ions (Pourrut et al. 2011).

Mercury Uptake

The availability of mercury in soil to plants is low. When absorbed, mercury tends to accumulate in roots, which indicate that the roots serve as a barrier to mercury translocation. Mercury concentration in aboveground parts of plants depends largely on foliar uptake of Hg^0 volatilized from the soil. Uptake of mercury has been found to be plant specific in bryophytes, lichens, wetland plants, woody plants, and crop plants. Factors that affect uptake of mercury by plants include organic matter content, cation exchange capacity, oxide and carbonate content, redox potential, and total metal content of soil. With lower levels of mercury pollution, the amounts in crops are below the permissible levels. Mercury concentrations in the plants (stems and leaves) are always greater when the metal is introduced in organic form. Mercury-vapor uptake by leaves of the C3 species oats, barley, and wheat is five times greater than that by leaves of the C4 species corn, sorghum, and crabgrass. Such differential uptake by C3 and C4 species is largely attributable to internal resistance to mercury-vapor binding. Airborne mercury thus seems to contribute significantly to the mercury content of crops and thereby to its intake by humans as food (Patra and Sharma 2000).

Nickel Uptake

Plants absorb Ni through the roots by passive diffusion and active transport (Seregin and Kozhevnikova 2006). The ratio of uptake between active and passive transport varies with the species, form of Ni, and concentration in the soil (Vogel-Mikus et al. 2005). The overall uptake of Ni by plants depends on the concentration of Ni^{2+}, plant metabolism, the acidity of soil or solution, the presence of other metals, and organic matter composition (Chen et al. 2009). However, uptake of Ni usually declines at higher pH values of the soil solution due to the formation of less soluble complexes (Temp 1991). For example, the uptake of Ni^{2+} by *Lathyrus sativus* reportedly increased with increasing pH up to 5.0 and decreased as the pH is increased further up to 8.0 (Pandaa et al. 2007). Moreover, Ni^{2+} ion may also compete with other essential metal ions when it is absorbed by roots. The uptake of heavy metals from the soil solution is strongly affected by calcium ion. Ca^{2+} lowered the absorption of Ni^{2+} in *Arabidopsis bertolonii*, an endemic plant of serpentine soils, but promoted Ni^{2+} absorption in *Berkheya coddii* (Boyd and Martens 1998). The inhibitory effect of various metal ions on absorption and translocation of Ni^{2+} from roots to shoots varied as $Fe^{3+} > Co^{2+} > Ca^{2+} > Mg^{2+} > NH_4^+ > K^+ > Na^+$ (Temp 1991). Besides being absorbed by roots, Ni can also enter into the plants via leaves. When a radioisotope of ^{63}Ni was applied on the leaves of *Helianthus annuus*, 37 % of the total amount was translocated to other plant organs (Sajwan et al. 1996). Similar trend was also observed when oat, soybean, tomato, and eggplant leaves were sprayed with Ni salt solution (Hirai et al. 1993). The path of Ni transport in plants is from root to shoot and makes an exit through transpiration stream (Neumann and Chamel 1986) via the xylem.

6.2.12.4 Heavy Metals in Plant Food

The levels of heavy metals in edible plant parts collected from crop fields and markets of vegetables, cereals, fruits, and spices have considerably been investigated. Metal contamination of plant materials have been reported from metal-contaminated places such as mining and industrial areas and in crop fields amended with sludge-based fertilizers or irrigated with contaminated water.

Lead concentration in polished rice from paddy grown in a contaminated site of China reached 0.69 mg kg^{-1}, which is higher than the maximum allowable concentration by the safety criteria for milled rice for China (0.20 mg kg^{-1}). Cadmium contents in 31 % of the rice samples exceeded the national maximum allowable concentration (Fu et al. 2008). Levels of Cd, Zn, Pb, and Cr in roots, stems, and leaves of the vegetable plants *Amaranthus viridis* and *Talinum triangulare* planted on poultry dumpsite of Nigeria were 0.62–2.74, 50.67–102.98, 2.27–7.21, and 0.64–4.45 mg kg^{-1}, respectively. Some of these values were above the safe levels (Adefila et al. 2010). Metal levels in vegetables, including parsley, onion, lettuce, garlic, peppermint, spinach, broad bean, chard, purslane, grapevine leaves, and fruits, including tomato, cherry, grape, and strawberry grown in Manisa region of Turkey, ranged from 0.56 to 329.7, 0.01 to 5.67, 0.26 to 30.68, 0.001 to 0.97, and 0 to 0.06 mg kg^{-1} for Fe, Cu, Zn, Pb, and Cd, respectively (Bagdatlioglu et al. 2010). Contamination with heavy metals of fruits including orange, mango, almond, lemon, sweet orange, grapefruits, chiku, papaya, muskmelon, apple, quince, grape, strawberry, banana, pineapple, papaya, carambola, longan, wampee, date palm, and apricot have been reported (Davarynejad et al. 2010). Concentrations of Cu, Ni, Zn, Fe, Pb, and Hg of 15 common spices available at local markets in the Kumasi Metropolis of Ghana ranged from 9 to 21, 31 to 50, 59 to 74, 110 to 494, 96 to 115, and 0.001 to 0.025 mg kg^{-1}, respectively (Nkansan and Amoako 2010). Permissible limits of heavy metals in edible parts of plants are shown in Table 6.8.

6.2.12.5 Toxicity of Heavy Metals

Organisms are exposed to soil contaminants by direct contact, handling, absorption, or ingestion and food intake. Heavy metals are bioaccumulated by incorporating into tissues, and as they are transferred through the food chain, they are biomagnified at higher trophic levels.

Effect of Heavy Metals on Soil Microorganisms

Some heavy metals are essential micronutrients (cobalt, chromium, nickel, iron, manganese, and zinc) for soil microorganisms (Bruins et al. 2000). They are involved in redox processes, in order to stabilize molecules through electrostatic interactions, as catalysts in enzymatic reactions, and regulating the osmotic balance (Hussein et al. 2005). On the other hand, some other heavy metals (cadmium,

Table 6.8 The permissible limits of some heavy metals in plant food

Metal	Plant food material	Permissible limit (mg kg^{-1})	References
Copper	All food	10.00	CAC (1993)
Cadmium	All food	0.05	Walker (1988)
	Vegetables and fruits, excluding leafy vegetables, fresh herbs, mushrooms	0.05	Graffham (2006)
	Leafy vegetables, fresh herbs, celeriac, and all cultivated fungi	0.20	Graffham (2006)
	Stem vegetables, root vegetables, and potatoes (peeled)	0.10	Graffham (2006)
Lead	Cereals and legumes	0.20	CAC (2003)
	Vegetables, excluding brassica, leafy vegetables, fresh herbs, and all fungi	0.10	Graffham (2006)
	For potatoes the maximum level applies to peeled potatoes		
	Brassica, leafy vegetables, and all cultivated fungi	0.30	Graffham (2006)
Zinc	Grains	50.00	USDA (2003)
	Beans	100.00	

WHO permissible limits in spices for Cu, Ni, Zn, Fe, Pb, and Hg are 50, 50, 100, 300, 100, and 10 mg kg^{-1}, respectively (Nkansan and Amoako 2010)

mercury, and lead) have no biological role and are detrimental to the organisms even at very low concentrations. High levels of both of the essential and nonessential metals are toxic to the soil microorganisms (Pawloska and Charvat 2004). Heavy metals adversely affect growth, morphology, and biochemical activities of microorganisms and ultimately reduce their biomass and diversity (Roane and Pepper 2000). Heavy metals can damage the cell membranes, alter enzymes specificity, disrupt cellular functions, and damage the structure of the DNA. Toxicity of these heavy metals occurs through the displacement of essential metals from their native binding sites or through ligand interactions (Bruins et al. 2000). Also, toxicity can occur as a result of alterations in the conformational structure of the nucleic acids and proteins and interference with oxidative phosphorylation and osmotic balance (Bruins et al. 2000). Heavy metals bring about change of the diversity, population size, and overall activity of the soil microbial communities (Kelly et al. 2003). Leita et al. (1995) studied influence of Pb, Cd, and Ti on microbial biomass, survival, and activity during a laboratory incubation of soil. In comparison to uncontaminated soil, the microbial biomass C decreased sharply in soil contaminated with Cd and Ti, whereas the addition of Pb did not have any significant inhibitory effect on the level of microbial biomass C. Long-term heavy metal contamination of soils reduces microbial respiration (Doelman and Haanstra 1984). Many reports have shown large reductions in microbial activity due to short-term exposure to toxic metals too (Hemida et al. 1997). Bacterial activity, measured by thymidine incorporation technique, had been shown to be very sensitive to metal pollution both under laboratory and field conditions (Diaz-Ravina and Baath 1996). The size of mycorrhizal roots has been found to decrease in soils containing high concentrations of heavy metals such as Cu, Pb, and Zn (Bell et al. 1988). Wastewaters containing high concentrations of heavy metals have also been shown to inhibit mycorrhizal infection of soybean, especially in soils with low pH (Angle and Heckman 1986). Lead and copper are less mobile than Zn and Cd. Ni, Cd, and Zn are potentially more serious contaminants of soil solutions than Cu and Pb (Biddapa et al. 1982). Absence of nitrogen fixation in clover crops grown on soils contaminated with heavy metals over a long period of time has been found to result from a survival only of ineffective rhizobial strains (Giller et al. 1989).

Toxicity of Heavy Metals to Plants

Plants absorb heavy metals, essential or nonessential, from soils. Copper, molybdenum, zinc, and nickel are essential trace elements required by plants in extremely small amounts. If absorbed in relatively large amounts, all essential or nonessential heavy metals become toxic to plants. According to Dan et al. (2008), heavy metals are potentially toxic for plants; phytotoxicity results in chlorosis, weak plant growth, yield depression, reduced nutrient uptake, disorders in plant metabolism, and, in leguminous plants, a reduced ability to fix molecular nitrogen. Metals inhibit seed germination, seedling growth, photosynthesis, and enzyme activity, but the effects vary with the metals, their concentrations, as well as speciation and the plant species. In a study Fargasova (1994) observed that Cr, Cd, Hg, Pb, and As all reduced seed germination of mustard (*Sinapis alba*), but As was the most inhibitory. Cadmium was found to be less toxic for germination of *Sinapis alba* seeds, but it is highly toxic to mung bean (*Vigna radiata*) seeds. Spinach, soybean, and curly cress, for instance, were sensitive to Cd, whereas cabbage and tomato were resistant. Toxicity symptoms of some heavy metals to plants are summarized in Table 6.9.

Heavy Metal Pollution and Earthworms

Earthworms are wonderful creatures of the soil. They constitute the largest terrestrial faunal biomass. They live in soil, they modify soil, they ingest soil, and they enrich soil. But they themselves are affected by the adverse conditions of soil. For example, earthworms can be exposed to elevated metal levels in soil by direct dermal contact or by ingestion of pore water, polluted food, and ingested soil particles (Lanno et al. 2004). Saxe et al. (2001) estimated that earthworms *Eisenia andrei* uptake more than 96 % of Cd and Cu and 82 % of Zn by dermal contact. Hobbelen et al. (2006) observed bioaccumulation of Cd, Cu, and Zn by the earthworms *Lumbricus rubellus* and *Aporrectodea caliginosa* in soils with high binding capacity. In the transfer of pollutants towards other trophic levels, earthworms occupy a key position (Granval and Aliaga 1988). Earthworms were found to have a high potential for Cd accumulation in polluted floodplains (Hendriks et al. 1995). They have been considered useful for assessing heavy metal pollution in soils (Menzie et al. 1992) because earthworm biomass and abundance were found to be more sensitive to pollution in comparison with other indicator taxa (Spurgeon et al. 1996).

Table 6.9 Toxicities of heavy metals in plants

Heavy metals	Common toxicities of heavy metals on plants
Arsenic	Detrimental to mung bean seed germination
Cadmium	Inhibition of the biochemical processes – photophosphorylation, ATP synthesis, mitochondrial NADH oxidation, and electron-transport system; it affects seed germination and seedling growth. Rice plants are particularly vulnerable
Chromium	Toxic at a tissue concentration of ~0.1 mmol kg^{-1}; toxic effects mainly on roots and a direct or indirect effect on leaves resulting in intense growth inhibition; symptoms of Cr toxicity appear as severe wilting and chlorosis of plants
Copper	Concentrations of <4 ppm are deficient, and >20 ppm are toxic. Cu is a constituent of a number of plant enzymes, which trigger a variety of physiological processes in plants such as photosynthesis, respiration, cell wall metabolisms, and seed formation
Lead	Inhibition of seed germination and seedling growth
Mercury	Toxic effects of Hg on plants include retardation of growth and premature senescence. Hg has been shown to inhibit synthesis of proteins in plant leaves and to reduce photosynthetic activity as it has a strong affinity for sulfhydryl or thiol groups, which are involved in enzymatic reactions. Hg also impairs cell division
Molybdenum	Mo is an important component of several enzymes that catalyze unrelated reactions. In plants, the most important functions of Mo are associated with N metabolism, e.g., with nitrogenase and nitrate reductase enzymes. Mo toxicity in plants has not been observed under field conditions
Nickel	Ni toxicity causes pale yellow stripes longitudinally in leaf. In extreme cases, the entire plant may turn white with marginal necrosis (burn). In dicots, Ni toxicity causes an interveinal chlorosis that looks very similar to manganese deficiency

Toxicity of Heavy Metals to Human

Toxicity of heavy metals involves either compounds with intrinsic toxicity or activated metabolites. It may be acute or chronic depending on the type of exposure. Changes in physiological and biochemical processes may be manifested in impaired central nervous system (CNS) function and oxidative metabolism, injury to the reproductive system, or altered DNA leading to carcinogenesis. Heavy metals can induce adverse effects in human by disruption or destruction of cellular structure, chemical combination with a cell constituent, inhibition of enzymes, initiation of a secondary action, free-radical-mediated reactions and disruption of reproductive function (Yu 2005). Toxicities due to most common heavy metal contaminants are listed in Table 6.10.

6.2.12.6 Heavy Metal Remediation of Soil

A variety of methods are employed for heavy metal remediation of soils. For the sake of brevity, a brief description of the methods is given in the following sections.

Table 6.10 Toxicities of heavy metals to human

Heavy metals	Common toxicities of heavy metals in human
Arsenic[a]	Arsenicosis is the most common disease caused by drinking As-contaminated water for a long time. Inorganic arsenic compounds are "known to be human carcinogens." Arsenic may cause cancers of the bladder, kidney, skin, liver, lung, colon, and lymph. Darkening of the skin and the appearance of small "corns" or "warts" on the palms, soles, and torso
Cadmium[a]	The well known itai-itai disease was caused by the consumption of rice grown by irrigating with Cd-contaminated river water in Japan after 1945. Cd toxicity causes renal tubular dysfunction, high blood pressure, lung damage, and lung cancer. Acute Cd inhalation may lead to pneumonitis and pulmonary edema. Chronic exposure via inhalation may cause emphysema. One of the most widely known toxic effects manifested by Cd poisoning is nephrotoxicity
Chromium[b]	Impaired growth, altered immune function, disturbances in aortic plaque and size, corneal lesion formation, and decrease in reproductive functions; both Cr(III) and Cr(VI) are potent human carcinogens. The major target organ for Cr(III) and Cr(VI) is the respiratory tract. Classical symptoms are perforations and ulcerations of the septum, bronchitis, decreased pulmonary function, and pneumonia
Copper[b]	Copper is an essential element. Cu toxicity in humans is very rare and is usually associated with long-term intake of cow's milk or with severe malnutrition in infants and young children
Lead[a]	Lead damages the central nervous system, kidneys, and lungs. It causes anemia; nausea, anorexia, and abdominal cramps; muscle aches and joint pain; difficulty in breathing, asthma, bronchitis, and pneumonia; and damage to the fetus and miscarriage. Children are more vulnerable to Pb poisoning than adults. Pb may cause male infertility
Mercury[a]	Severe mercury poisoning occurred in Minamata of Japan from contaminated fish and in Iraq from contaminated wheat. Wheat was sprayed with MeHg fungicide. In the outbreak more than 6,000 children and adults had been poisoned, with nearly 500 deaths. Symptoms include paresthesia, ataxia, dysarthria, and deafness. Chronic Hg poisoning may result from exposure to small amounts of Hg over long periods; symptoms include salivation, loss of appetite, anemia, gingivitis, excessive irritation of tissues, nutritional disturbances, and renal damage accompanied by proteinuria. Exposure to Hg may cause nausea, severe gastrointestinal irritation, and abdominal pain
Molybdenum[b]	Toxic effects of Mo include increased blood xanthine oxidase, increased concentrations of uric acid in blood and urine, and a high incidence of gout
Nickel[a]	Exposure to nickel sulfate and nickel chloride can cause vomiting and headaches and the metal can cross human placental barrier, affecting the fetus. Ni-sensitive individuals often develop contact dermatitis. Inhalation of Ni compounds has been considered responsible for lung, sinonasal, and laryngeal carcinomas. Other illnesses include pneumonoconiosis and emphysema. Ni may cause kidney cancer. Among different types of Ni compounds, crystalline nickel subsulfide is carcinogenic, whereas amorphous NiS is not

(continued)

Table 6.10 (continued)

Heavy metals	Common toxicities of heavy metals in human
Selenium[b]	It can cause nausea, vomiting, and diarrhea. Chronic oral exposure to high concentrations can produce selenosis. Major signs of selenosis are hair loss, nail brittleness, and neurological abnormalities. Brief exposures to high levels in air can result in respiratory tract irritation, bronchitis, difficulty breathing, and stomach pains. Longer-term exposure can cause respiratory irritation, bronchial spasms, and coughing
Zinc[c]	Zinc is considered to be relatively nontoxic, especially if taken orally. However, excess amount can cause system dysfunctions that result in impairment of growth and reproduction. The clinical signs of zinc toxicosis have been reported as vomiting, diarrhea, bloody urine, icterus, liver failure, kidney failure, and anemia (Fosmire 1990)

[a] Yu (2005)
[b] Bradl (2005)
[c] Duruibe et al. (2007)

There are some good reviews of the heavy metal remediation techniques (USEPA 1997; Stegmann et al. 2001; Bradl and Xenidis 2005; Wang et al. 2010). However, the choice of the method depends on the degree and kind of contamination, contaminating metal(s), cost involved, farm facilities, technology available, etc. As mentioned in relation to organic pollutant remediation, the methods may be on-site or off-site, in situ or ex situ, and physical, chemical, and biological. Physical methods include soil washing, encapsulation, vitrification, and electrokinesis; chemical methods include solidification, precipitation, and ion exchange; and biological methods use microorganisms or plants to remove heavy metals.

Physical Remediation

Soil Washing

Soil washing is usually done as an ex situ remediation process. Soil washing employs physical and/or chemical procedures to extract metal contaminants from soils. During soil washing, (1) those soil particles which host the majority of the contamination are separated from the bulk soil fractions, (2) contaminants are removed from the soil by aqueous chemicals and recovered from solution on a solid substrate, or (3) a combination of both (Dermont et al. 2008). The separated contaminants are disposed in landfill or further treated by chemical, thermal, or biological processes. By removing the majority of the contamination from the soil, the bulk fraction that remains can be (1) back filled, (2) used on another site as fill, or (3) disposed of relatively cheaply as nonhazardous material. Soil washing is a popular practice because it (1) completely removes the contaminants and hence ensures the rapid cleanup of a contaminated site, (2) meets specific criteria, (3) reduces or eliminates long-term liability, (4) may be the most cost-effective, and (5) may produce recyclable material or energy (GOC 2003). The disadvantages include the fact that the contaminants are simply moved to a different place, where they must be monitored, the risk of spreading contaminated soil and dust particles during removal and transport of contaminated soil, and the relatively high cost. For different heavy metals, extracting solutions for soil washing may be different. Several classes of chemicals used for soil washing include surfactants, cosolvents, cyclodextrins, chelating agents, and organic acids.

Soil Flushing

In soil flushing, an aqueous solution is injected into the contaminated zone of soil followed by extraction of groundwater and elutriate (flushing solution mixed with the contaminants) and aboveground treatment and discharge. The goal of this in situ flushing is to enhance the solubility or mobility of contaminants and accelerating the remediation process. Flushing solutions may include water or surfactants, cosolvents, acids, bases, oxidants, chelants, and solvents which percolate through the soil, and soluble compounds present in the soil are dissolved. The elutriate is pumped out of the contaminated zone into a water treatment system to remove pollutants.

Encapsulation

The basic principle is the underground construction of an impermeable vertical barrier to allow the containment of gases and liquids. A variety of construction methods such as cutoff slurry walls using mainly cement–bentonite–water slurries, thin walls, sheet pile walls, bored-pile cutoff walls, jet grouting curtains, injection walls, and frozen barriers have been developed. Encapsulation is recommended as the easiest way to safely dispose of metal polluted soils and hazardous wastes including hospital wastes. Contaminated soils and wastes are filled in leakproof containers. When the container is three-quarters full, a material such as cement (mortar), plastic foam, or clay is poured into the container until completely filled. After the material has hardened, the container is sealed and may be landfilled, stored, or buried. It is also possible to encapsulate chemical or pharmaceutical waste together with sharps (WHO 1999).

Solidification/Stabilization

Solidification is done through the addition of binding agents to a contaminated material to impart physical stability to contain contaminants in a solid product. Stabilization (fixation) involves the addition of reagents to the contaminated soil to produce more chemically stable constituents. The general approach for solidification/stabilization treatment processes involves mixing or injecting treatment agents to the contaminated soils. Inorganic binders such as clay (bentonite and kaolinite), cement, fly ash, blast furnace slag, calcium carbonate, Fe/Mn oxides, charcoal, zeolite, and organic stabilizers such as bitumen, composts, and manures, or a combination of organic–inorganic amendments may be used. The dominant mechanism by which metals are immobilized is by precipitation of hydroxides within the solid matrix

(Shively et al. 1986). Solidification/stabilization technologies are not useful for some forms of metal contamination, such as species that exist as oxyanions (e.g., $Cr_2O_7^{2-}$, AsO_3^-) or metals that do not have low-solubility hydroxides (e.g., Hg). Solidification/stabilization may not be applicable soils contaminated with organic pollutants. Cement-based binders and stabilizers are common materials used for implementation of solidification/stabilization technologies (Conner 1990). Portland cement, a mixture of Ca silicates, aluminates, aluminoferrites, and sulfates, is an important cement-based material. Pozzolanic materials, which consist of small spherical particles formed by coal combustion (such as fly ash) and in lime and cement kilns, are also commonly used for the purpose. Organic binders may also be used to treat metals through polymer microencapsulation. This process uses organic materials such as bitumen, polyethylene, paraffins, waxes, and other polyolefins as thermoplastic or thermosetting resins.

Vitrification

Vitrification involves the conversion of contaminated soil materials into glass-like substances. This is achieved by high-temperature treatment of the contaminated area that results in the formation of vitreous material. Most soils can be treated by vitrification, and a wide variety of inorganic and organic contaminants can be targeted. Vitrification may be performed ex situ or in situ although in situ processes are preferred due to the lower energy requirements and cost (USEPA 1992b). Typical stages in ex situ vitrification processes may include excavation, pretreatment, mixing, feeding, melting and vitrification, off-gas collection and treatment, and forming or casting of the melted product. Some additives such as sand, clay, and/or native soil may be used for vitrification. The vitrified waste may be recycled and used as clean fill, aggregate, or other reusable materials. In situ vitrification involves passing electric current through the soil using an array of electrodes inserted vertically into the contaminated region. Resistance heating in the starter path melts the soil. A single melt can treat up to 1,000 t of contaminated soil to depths of 20 ft, at a typical treatment rate of 3–6 t h^{-1}. Larger areas are treated by fusing together multiple individual vitrification zones (Wuana and Okieimen 2011).

Electrokinesis

Electrokinetic remediation offers a great potential for the remediation of hazardous waste sites, especially those containing fine-grained soils contaminated with heavy metals. It can be used either in situ or ex situ involving the application of a low level DC current or voltage gradient across electrodes that encompass the contaminated soil. As a result, the contaminants are transported toward either the cathode well/reservoir or the anode well/reservoir, depending on their charge. Numerous studies dealing with the electrokinetic remediation of soils contaminated with cationic metallic contaminants such as lead, copper, and cadmium have been reported (Hicks and Tondorf 1994). These studies have demonstrated migration of significant amounts of cationic metallic contaminants toward the cathode region where they are precipitated due to alkaline conditions. In addition, the introduction of weak acids such as acetic acid into the cathode well/reservoir can lower the soil pH near the cathode regions, thereby preventing precipitation of the contaminants. The contaminants are allowed to transport into the cathode well/reservoir for subsequent removal by either electroplating or pumping aboveground.

Chemical Remediation

Heavy metals occur as dissolved ions or bound to colloidal particles in soils. They are either precipitated or flocculated and separated. Precipitation is a process where ionic components transform into a non-soluble phase. In flocculation, small undissolved solids of colloidal size are aggregated into larger solid flocks. Precipitates and flocculates are then separated mechanically from the fluid by sedimentation, centrifugation, or flotation.

Precipitation

The most important chemical parameters in the precipitation process are pH and concentration of metal ions. Generally, heavy metals tend to be present in ionic form at low pH levels and precipitate when pH is raised. Heavy metals can be precipitated as insoluble hydroxides, sulfides, carbonates, and others by adding precipitants (digested sludge, Fe salts, calcium hydroxide, Al salts, etc.). They are then separated by different solid/liquid separation techniques. The precipitation of metal sulfides has been proven to be very effective. Adsorptive materials such as iron oxides, clay minerals, zeolites, and organic matter are sometimes used to immobilize the metals in soil.

Ion Exchange

Ion exchange is a matrix or resin laden with dissociable counter ions. Common ion exchangers are made of interlaced polystyrene and polyacrylate or condensation resins made from phenol and formaldehyde. When a solution containing dissolved metal ions are passed through the resin column, the metal ions are exchanged with adsorbed ions on resin surfaces. For regeneration, acids such as HCl and H_2SO_4 and alkaline solution (NaOH) are used. Wastewater produced in the process may be treated for precipitation or flocculation of metal ions.

Chelation

Chelating agents such as ethylenediaminetetraacetic acid (EDTA), nitriloacetic acid (NTA), diethylenetriaminepentaacetic acid (DTPA), and *S*,*S-ethylenediaminedisuccinic* acid (EDDS) can form strong metal–ligand complexes and are thus highly effective in remediating heavy metal-contaminated soils (Norvell 1984; Elliott and Brown 1989; Kim and Ong 1998). Among these chelators, NTA is a Class II carcinogen and DTPA is identified as toxic and a potential carcinogen. EDTA continues to be explored extensively for soil remediation because of its ability to mobilize metal cations efficiently coupled with only a minor impact on the physical and chemical properties of the soil matrix (Lee and Marshall 2003). The extraction kinetics of copper, zinc, iron, and manganese from the contaminated sediment of the Clark Fork River in western Montana, USA, with disodium ethylenediaminetetraacetate (Na2EDTA) as the extraction agent, were investigated. The results showed the extraction process consisted of rapid extraction in the first minutes followed by much slower extraction for the remainder of the experiment. The rate of extraction, particularly in the rapid phase, demonstrated clear pH dependence: the lower the pH, the faster the extraction rate.

Bioremediation

In bioremediation, organisms are employed in extraction and removal of metals from the contaminated soil. Organisms include microorganisms and higher plants.

Microbial Remediation

Natural organisms, either indigenous or extraneous are the prime agents used for bioremediation of heavy metals in soil (Prescott et al. 2002). The organisms that are utilized vary, depending on the chemical nature of the polluting agents and are to be selected carefully as they only survive within a limited range of chemical contaminants (Dubey 2004). Since numerous types of pollutants are to be encountered in a contaminated site, diverse types of microorganisms are likely to be required for effective mediation (Watanabe et al. 2001). Bioremediation can occur naturally or through intervention processes (Agarwal 1998). Natural degradation of pollutants relies on indigenous microflora that is effective against specific contaminants and it usually occurs at a slow rate. With intervention processes, the rate of biodegradation is aided by encouraging growth of microorganisms, under optimized physicochemical conditions (Smith et al. 1998a, b).

Phytoremediation

Phytoremediation uses the ability of plants to adsorb, degrade, volatilize, or accumulate contaminants in soil, sediments, surface, or groundwater. According to Ghosh and Singh (2005), the chief processes of phytoremediation are phytostabilization, phytodegradation, phytoaccumulation, phytovolatilization, rhizodegradation, and evapotranspiration. Phytoaccumulation or phytoextraction is the most commonly and popularly used process of phytoremediation of heavy metals from contaminated soils. Phytoextraction utilizes the metal hyperaccumulating ability of some plants in their aboveground parts. Selected hyperaccumulating plants are grown in contaminated soils and harvested aboveground parts are treated in different ways (reduction in volume and weight, composting, compaction, burial, thermal treatment).

6.2.12.7 Metal Hyperaccumulator Plants

Some metal-tolerant plants are characterized by the ability to accumulate very high concentrations of metals, far in excess of normal physiological requirements (if any) and far in excess of the levels found in the majority of other species tolerant of metals. These plants are known as metal "hyperaccumulator" plants. The term hyperaccumulation was first applied by Jaffre et al. (1976) for nickel concentrations in the New Caledonian tree species *Sebertia acuminata*. According to Reeves (1992), a hyperaccumulator of Ni is a plant in which a Ni concentration of at least 1,000 mg kg^{-1} has been recorded in the dry matter of any aboveground tissue in at least one specimen growing in its natural habitat. Later, 1,000 mg kg^{-1} criterion was also taken for hyperaccumulation of Cu, Co, and Pb. A concentration >10,000 mg kg^{-1} is applied to Mn and Zn hyperaccumulation. Many plants have been identified as metal hyperaccumulators. However, a list of some important metal hyperaccumulator plants obtained from Reeves and Baker (2000) is given below.

Metal	Hyperaccumulator plants
Chromium	*Alyxia rubricaulis*, *Maytenus bureaviana*, *M. pancheriana*, *M. sebertiana*, *Garcinia amplexicaulis*, *Austromyrtus bidwillii*, *Eugenia clusioides*, *Eugenia* sp., *Beaupreopsis paniculata*, *Macadamia angustifolia*, *M. neurophylla*, *Astragalus stanleya*, *Haplopappus*, *Machaeranthera*
Cobalt and copper	*Pandiaka metallorum*, *Anisopappus davyi*, *Cyanotis longifolia*, *Ascolepis metallorum*, *Bulbostylis pseudoperennis*, *Phyllanthus williamioides*, *Crotalaria cobalticola*, *Vigna dolomitica*, *Aeollanthus subacaulis* var. *linearis*, *Haumaniastrum robertii*, *Eragrostis racemosa*, *Actiniopteris* sp., *Buchnera henriquesii*, *Sopubia neptunii*, *Triumfetta dekindtiana*, *T. welwitschii* var. *descampii*, *Xerophyta retinervis* var. *equisetoides*
Cadmium, lead and zinc	*Arabidopsis halleri*, *Thlaspi caerulescens*, *T. caerulescens*, *T. brachypetalum*, *T. caerulescens*, *T. ochroleucum*, *T. cepaeifolium*, *T. praecox*, *T. stenopterum*, *T. tatrensinuartia verna*, *Polycarpaea synandra*, *Dichapetalum gelonioides*, *Armeria maritima*, *Agrostis tenuis*, *Arrhenatherum elatius*, *Festuca ovina*, *Rumex acetosa*, *Viola calaminaria*
Manganese	*Vaccinium myrtillus*, *Austromyrtus bidwillii*
Nickel	*Berkheya coddii*, *Pentacalia* (10 species), *Senecio* (9 species), *Alyssum* (52 taxa), *Bornmuellera* (6 taxa), *Cochlearia aucheri*, *Peltaria emarginata*, *Streptanthus polygaloides*, *Thlaspi* (23 taxa)

A concentration higher than 100 mg kg^{-1} is taken for Cd hyperaccumulation (Reeves and Baker 2000). Some hyperaccumulator plants accumulate very high amounts of heavy metals. For example, *Thlaspi calaminare* and *Phyllanthus serpentinus* were reported to accumulate 39,600 mg kg^{-1} Zn and 38,100 mg kg^{-1} Ni, respectively, in their leaves (Siegel 2002).

Study Questions

1. What do you mean soil pollution? How do polluted soils affect water and air quality?

2. What are wastes? Discuss modes of disposal of wastes in the environment. List different soil pollutants and their sources.
3. Agrochemicals are a major source of soil pollution – explain. Give a list of pesticides that contain hazardous compounds. Discuss persistence of pesticides.
4. Discuss in situ and ex situ remediation of organic pollutants. Bioremediation of organic pollutants is environmentally more acceptable – discuss.
5. What are the environmentally important heavy metals? What are the effects of heavy metals on plants, humans, and microorganisms? Mention the methods of remediation of metal polluted soils.

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