4 THE HUMAN IMPACT ON THE SOIL

Introduction

Soil is material composed of mineral particles, voids filled by gases or solutes, and organic remains that overlies the bedrock or parent sediment and supports the growth of roots. Humans live close to and depend on the soil. It is one of the thinnest and most vulnerable human resources and is one upon which, both deliberately and inadvertently, humans have had major impacts. Moreover, such impacts can occur with great rapidity in response to land-use change, new technologies, or waves of colonization (see Russell and Isbell, 1986, for a review in the context of Australia).

Natural soil is the product of a whole range of factors and the classic expression of this is that of Jenny (1941):

$$s = f(cl, o, r, p, t \ldots)$$

where s denotes any soil property, cl is the regional climate, o the biota, r the topography, p the parent material, t the time (or period of soil formation), and the dots represent additional, unspecified factors. In reality soils are the product of highly complex interactions of many interdependent variables, and the soils themselves are not merely a passive and dependent factor in the environment. Nonetheless, following Jenny's subdivision of the classic factors of soil formation, one can see more clearly the effects humans have had on soil, be they detrimental or beneficial. These can be summarized as follows (adapted from the work of Bidwell and Hole, 1965):

1 *Parent material*
   Beneficial: adding mineral fertilizers; accumulating shells and bones; accumulating ash locally; removing excess amounts of substances such as salts.
   Detrimental: removing through harvest more plants and animal nutrients than are replaced; adding materials in amounts toxic to plants or animals; altering soil constituents in a way to depress plant growth.
2 *Topography*
   Beneficial: checking erosion through surface roughening, land forming and structure building; raising land level by accumulation of material; land leveling.
   Detrimental: causing subsidence by drainage of wetlands and by mining; accelerating erosion; excavating.

3 *Climate*
Beneficial: adding water by irrigation; rainmaking by seeding clouds; removing water by drainage; diverting winds, etc.
Detrimental: subjecting soil to excessive insolation, to extended frost action, to wind, etc.

4 *Organisms*
Beneficial: introducing and controlling populations of plants and animals; adding organic matter including 'night-soil'; loosening soil by plowing to admit more oxygen; fallowing; removing pathogenic organisms, e.g., by controlled burning.
Detrimental: removing plants and animals; reducing organic content of soil through burning, plowing, overgrazing, harvesting, etc.; adding or fostering pathogenic organisms; adding radioactive substances.

5 *Time*
Beneficial: rejuvenating the soil by adding fresh parent material or through exposure of local parent material by soil erosion; reclaiming land from under water.
Detrimental: degrading the soil by accelerated removal of nutrients from soil and vegetation cover; burying soil under solid fill or water.

Space precludes, however, that we can follow all these aspects of anthropogenic soil modification or, to use the terminology of Yaalon and Yaron (1966), of *metapedogenesis*. We will therefore concentrate on certain highly important changes which humans have brought about, especially chemical changes (such as salinization and lateritization), various structural changes (such as compaction), some hydrological changes (including the effects of drainage and the factors leading to peat-bog development), and, perhaps most important of all, soil erosion.

Salinity: natural sources

Increasing salinity has a whole series of consequences that include a reduction in the availability of potable water (for humans and/or their stock), deterioration in soil structure, reduction in crop yields, and decay of engineering structures and cultural treasures. It is, therefore, a major environmental issue.

Many semi-arid and arid areas are, however, naturally salty. By definition they are areas of substantial water deficit where evapotranspiration exceeds precipitation. Thus, whereas in humid areas there is sufficient water to percolate through the soil and to leach soluble materials from the soil and the rocks into the rivers and hence into the sea, in deserts this is not the case. Salts therefore tend to accumulate. This tendency is exacerbated by the fact that many desert areas are characterized by closed drainage basins, which act as terminal evaporative sumps for rivers.

The amount of natural salinity varies according to numerous factors, one of which is the source of salts. Some of the salts are brought into the deserts by rivers. A second source of salts is the atmosphere – a source that in the past has often been accorded insufficient importance. Rainfall, coastal fogs, and **dust storms** all transport significant quantities of soluble salts. Further soluble salts may be derived from the weathering and solution of bedrock. In the Middle East, for example, there are extensive salt domes and evaporite beds within the bedrock, which create locally high groundwater and surface-water salinity levels. In other areas, such as the Rift Valley of East Africa, volcanic rocks may provide a large source of sodium carbonate to groundwater, while elsewhere the rocks in which groundwater occurs may contain salt because they are themselves ancient desert sediments. Even in the absence of such localized sources of highly saline groundwater it needs to be remembered that over a period of time most rocks will provide soluble products to groundwater, and in a closed hydrological system such salts will eventually accumulate to significant levels.

A further source of salinity may be marine transgressions. At times of higher sea levels, it has sometimes been proposed (see, e.g., Godbole, 1972) that salts would have been laid down by the sea. Likewise in coastal areas, salts in groundwater aquifers may be contaminated by contact with seawater.

Human agency and increased salinity

Human activities cause enhanced or secondary salinization in drylands in a variety of ways (Goudie and Viles, 1997). In Table 4.1 these mechanisms are grouped into five main classes: irrigation salinity; dryland salinity; urban salinity; salinity brought about by interbasin water transfers; and coastal zone salinity.

Human-induced salinization affects about 77 million hectares on a global basis, of which 48 million hectares are in susceptible drylands (Middleton and Thomas, 1997) (Table 4.2).

Table 4.1 Enhanced salinization

Type	*Cause*
Irrigation salinity	Rise in groundwater Evaporation of water from fields Evaporation of water from canals and reservoirs Waterlogging produced by seepage losses
Dryland salinity	Vegetation clearance
Urban salinity	Water importation and irrigation Faulty drains and sewers
Interbasin water transfers	Mineralization of lake waters Deflation of salts from desiccating lakes
Coastal zone salinity	Overpumping Reduced freshwater recharge Sea-level rise Ground subsidence

Table 4.2 Global extent of human-induced salinization in the susceptible drylands (million ha). Source: GLASOD; Middleton and Thomas (1997, table 4.17)

Continent	*Light*	*Moderate*	*Strong*	*Extreme*	*Total*
Africa	3.3	1.9	0.6	–	5.8
Asia	10.7	8.1	16.2	0.4	35.4
South America	0.9	0.1	–	–	1.0
North America	0.3	1.2	0.3	–	1.8
Europe	0.8	1.7	0.5	–	3.0
Australasia	–	0.5	–	0.4	0.9
Global total	16.0	13.5	17.6	0.8	47.9

Irrigation salinity

In recent decades there has been a rapid and substantial spread of irrigation across the world (Table 4.3). The irrigated area in 1900 amounted to less than 50 million hectares. By 2000 the total area amounted to five times that figure. During the 1950s the irrigated area was increasing at over 4% annually, though this figure has now dropped to only about 1%. This spread of irrigation has brought about a great deal of salinization and waterlogging (Figures 4.1 and 4.2).

The amount of salinized irrigated land varies from area to area (Table 4.4), but in general ranges between 10 and 50% of the total. However, there is a considerable range in these values according to the source of

Table 4.3 Estimates of the increasing area of irrigated land on a global basis. Source: Goudie and Viles (1997, table 1.3)

Year	*Irrigated area (10^6 ha)*
1900	44–48
1930	80
1950	94
1955	120
1960	168
1980	211
1990 (estimate)	240

Figure 4.1 An irrigated field in southern Morocco. The application of large amounts of irrigation of water causes groundwater levels to rise and high air temperatures lead to rapid evaporation of the water. This leads to the eventual build up of salts in the soil.

Figure 4.2 The extension of irrigation in the Indus valley of Pakistan by means of large canals has caused widespread salination of the soils. Waterlogging is also prevalent. The white efflorescence of salt in the fields has been termed 'a satanic mockery of snow'.

Table 4.4 Salinization of irrigated cropland. Source: FAQ data as summarized in World Resources Institute (1988, table 19.3)

Country	*Percentage of irrigated lands affected by salinization*
Algeria	10–15
Australia	15–20
China	15
Colombia	20
Cyprus	25
Egypt	30–40
Greece	7
India	27
Iran	< 30
Iraq	50
Israel	13
Jordan	16
Pakistan	< 40
Peru	12
Portugal	10–15
Senegal	10–15
Sri Lanka	13
Spain	10–15
Sudan	< 20
Syria	30–35
USA	20–25

the data (compare Table 4.5) and this may in part reflect differences in the definition of the terms 'salinization' and 'waterlogging' (see Thomas and Middleton, 1993).

Irrigation causes secondary salinization in a variety of ways (Rhoades, 1990). First, the application of irrigation water to the soil leads to a rise in the water table so that it may become near enough to the ground surface for capillary rise and subsequent evaporative concentration to take place. When groundwater comes within 3 m of the surface in clay soils, and even less for silty and sandy soils, capillary forces bring moisture to the surface where evaporation occurs. There is plenty of evidence that irrigation does indeed lead to rapid and substantial rises in the position of the water table. Rates typically range between 0.2 and 3 m per year.

Second, many irrigation schemes, being in areas of high temperatures and rates of evaporation, suffer from the fact that the water applied over the soil surface is readily concentrated in terms of any dissolved salts it may contain. This is especially true for crops with a high water demand (e.g., rice) or in areas where, for one reason or another, farmers are profligate in their application of water.

Third, the construction of large dams and barrages creates extensive water bodies from which further evaporation can take place, once again leading to the concentration of dissolved solutes.

Fourth, notably in sandy soils with high permeability, water seeps both laterally and downwards from irrigation canals so that waterlogging may occur. Many irrigation canals are not lined, with the consequence that substantial water losses can result.

Table 4.5 Global estimate of secondary salinization in the world's irrigated lands. Source: Ghassemi et al. (1995, table 18). Reproduced by permission of CAB International and the University of New South Wales Press

Country	*Cropped area (Mha)*	*Irrigated area (Mha)*	*Share of irrigated to cropped area (%)*	*Salt-affected land in irrigated area (%)*	*Share of salt-affected to irrigated land (%)*
China	96.97	44.83	46.2	6.70	15.0
India	168.99	42.10	24.9	7.00	16.6
CIS	232.57	20.48	8.8	3.70	18.1
USA	189.91	18.10	9.5	4.16	23.0
Pakistan	20.76	16.08	77.5	4.22	26.2
Iran	14.83	5.74	38.7	1.72	30.0
Thailand	20.05	4.00	19.9	0.40	10.0
Egypt	2.69	2.69	100.00	0.88	33.0
Australia	47.11	1.83	3.9	0.16	8.7
Argentina	35.75	1.72	4.8	0.58	33.7
South Africa	13.17	1.13	8.6	0.10	8.9
Subtotal	852.80	158.70	18.8	29.62	20.0
World	1473.70	227.11	15.4	45.4	20.0

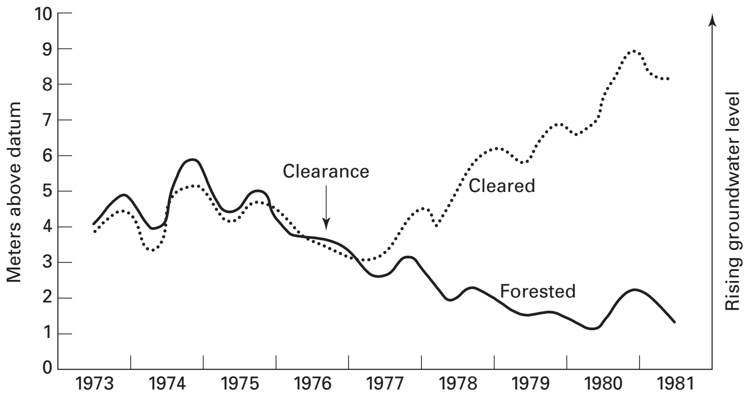

Figure 4.3 Comparison of hydrographs recorded from the boreholes in Wights (———) and Salmon (------) catchments in Western Australia. Both catchments were forested until late in 1976 when Wights was cleared (modified after Peck, 1983, figure 1).

Dryland salinity

A prime cause of dryland salinity extension is vegetation clearance (Peck and Halton, 2003). The removal of native forest, by reducing interception and evapotranspirational losses, allows a greater penetration of rainfall into deeper soil layers which causes groundwater levels to rise, thereby creating conditions for the seepage of sometimes saline water into low-lying areas. This is a particularly serious problem in the wheat belt of Western Australia and in some of the prairies of North America. In the case of the former area it is the clearance of *Eucalyptus* forest that has led to the increased rate of groundwater recharge and to the spreading salinity of streams and bottomlands. Salt 'scalds' have developed. The speed and extent of groundwater rise following such forest clearance is shown in Figure 4.3. Until late 1976 both the Wights and Salmon catchments were forested. Then the Wights was cleared. Before 1976 both catchments showed a similar pattern of groundwater fluctuation, but after that date there was a marked divergence of 5.7 m (Peck, 1983). The process can be reversed by afforestation (Bari and Schofield, 1992). Revegetation policies could also provide increased carbon sinks and so could provide synergistic value (Pittock and Wratt, 2001: 603).

Groundwater levels have increased some tens of meters since clearance of the natural vegetation began. They have increased by up to 30 m since the 1880s in southeastern Australia and by about 20 m in parts of southwest Australia. In some of the upland areas of New South Wales, groundwater levels have increased by up to 60 m over the past 70 to 80 years (http://www.agso.gov.au/information/structure/egg/mb/salinity2.html)

While in New South Wales, the area of land affected by dryland salinity is currently reported to be about 120,000 hectares. However, if current land-use trends and groundwater rise continue, this figure has the potential to increase to as much as 7.5 million hectares by 2050 (http://nccnsw.org.au/veg/context/salinity_fs.html). In Western Australia there is already an estimated 1.8 million hectares of farmland that is salt-affected. This area could double in the next 15 to 20 years, and then double again (http://www.agric.wa.gov.au/progserv/natural/trees/salinity/salwa.html) before reaching equilibrium. In all, some 6.1 million hectares have the potential to be affected by dryland salinity.

Dryland salinity is also a major problem on the Canadian prairies. In Alberta, approximately 0.65 million hectares are affected by secondary salinity, with an average crop yield reduction of 25%. In Saskatchewan 1.3 million hectares are affected, and in Manitoba 0.24 million hectares (http://www.agric.gov.ab.ca/sustain/soil/salinity/).

Rising water tables resulting from land-use changes are now being identified in other areas. For example, there has been a marked rise in the water table of the Continental Terminal in southwest Niger (Leduc et al., 2001). The rates have been between 0.01 and 0.45 million per year. The reason for this is the replacement of natural woodland savanna with millet fields and associated fallows. This has promoted increased surface runoff, which concentrates in temporary endoreic ponds and then infiltrates to the water table.

Urban salinity

Recent decades have seen a great growth in urban areas in drylands. This is, for example, the case with some of the Gulf States of the Middle East. The city of Abu Dhabi had a population of around 8000 in 1960. By 1984 it had reached 243,000, a 30-fold increase, and by 1995, 928,000, a 116-fold increase.

Urbanization can cause a rise in groundwater levels by affecting the amount of moisture lost by evapotranspiration. Many elements of urbanization, and in particular the spread of impermeable surfaces (roads, buildings, car parks, etc.), interrupt the soil evaporation process so that groundwater levels in *sabkha* (salt plain) areas along the coast of the Arabian Gulf rise at a rate of 40 cm per year until a new equilibrium condition is attained; the total rise from this cause may be 1–2 m (Shehata and Lotfi, 1993). This can require the construction of horizontal drains.

Urbanization can lead to other changes in groundwater conditions that can aggravate salinization. In some large desert cities the importation of water, its usage, wastage, and leakage can produce the ingredients to feed this phenomenon. This has, for example, been identified as a problem in Cairo and its immediate environs (Hawass, 1993). The very rapid expansion of Cairo's population has outstripped the development of an adequate municipal infrastructure. In particular, leakage losses from water pipes and sewers have led to a substantial rise in the groundwater level and have subjected many buildings to attack by sulfate- and chloride-rich water. There are other sites in Egypt where urbanization and associated changes in groundwater levels have been identified as a major cause of accelerated salt weathering of important monuments. Smith (1986: 510), for example, described the damage to tombs and monuments in Thebes and Luxor in the south of the country.

Interbasin water transfers

A further reason for increases in levels of salinity is the changing state of water bodies caused by interbasin water transfers. The most famous example of this is the shrinkage of the Aral Sea, the increase in its mineralization, and the deflation of saline materials from its desiccating floor and their subsequent deposition downwind. Some tens of millions of tonnes of salt are being translocated by dust storms (Figure 4.4) each year (Saiko and Zonn, 2000). The sea itself has had its mineral content increased more than threefold since 1960. Data on recent salinity enhancement in lakes from the USA, Asia, and Australia is provided by Williams (1999) (Table 4.6).

Another important illustration of the effects of interbasin water transfers is the desiccation of the Owens Lake in California. Diversion of water to feed the insatiable demands of Los Angeles has caused the lake to dry out, so that saline dust storms have become

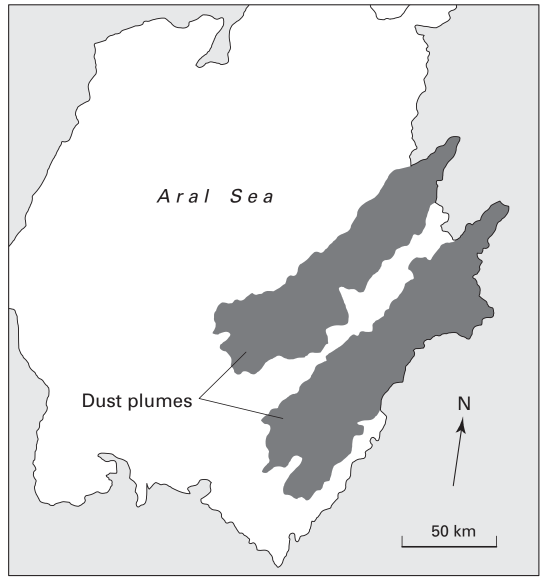

Figure 4.4 Dust plumes caused by the deflation of salty sediments from the drying floor of the Aral Sea as revealed by a satellite image (153/Metero-Priroda, 18 May 1975) (modified after Mainguet, 1995, figure 4).

Table 4.6 Enhanced salinity of lake basins. Source: Data in Williams (1999)

Lake	*Period*	*Change* ($g\ L^{-1}$)
Mono (California)	1941–1992	48 to 90
Pyramid Lake (Nevada)	1933–1980	3.75 to 5.5
Dead Sea	1910–1990s	200 to 300
Aral Sea	1960–1991	10 to 30
Qinghai (China)	1950s–1990s	6 to 12
Corangamite (Australia)	1960–1990s	35 to 50

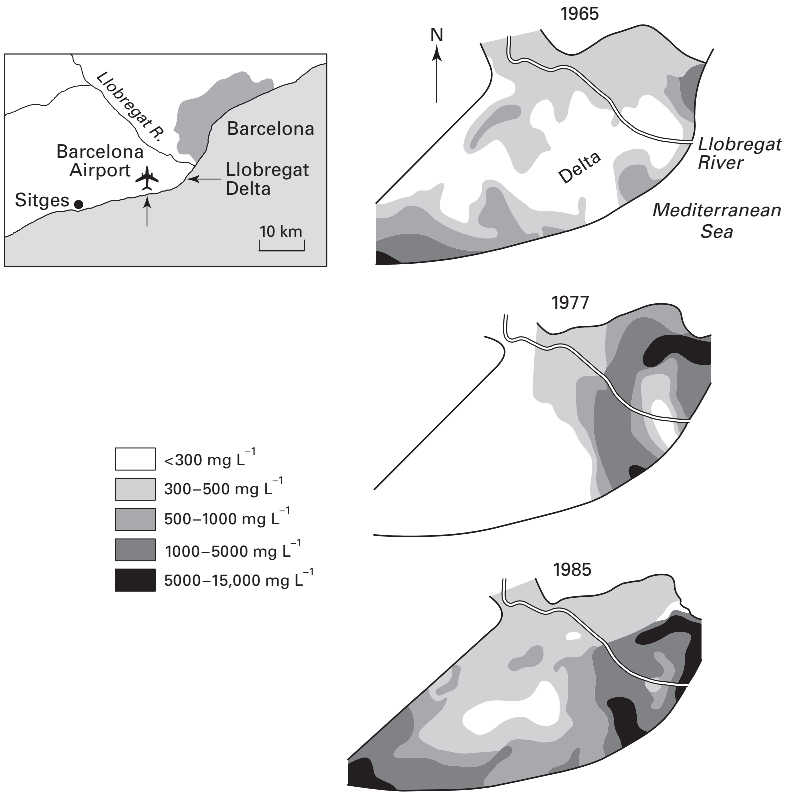

Figure 4.5 Changes in the chloride concentration of the Llobregat Delta aquifer, Barcelona, Spain as a result of seawater incursion caused by the overpumping of groundwater (modified from Custodio et al., 1986).

an increasingly serious issue (Gill, 1996). Sampling of airborne dust in the area has shown that 20–70% of the dust is soluble salts (Tyler et al., 1997).

Coastal zone salinity

Another prime cause of the spread of saline conditions is the incursion of seawater brought about by the overpumping of groundwater. Saltwater displaces less saline groundwater through a mechanism called the Ghyben–Herzberg principle. The problem presents itself on the coastal plain of Israel, in parts of California, on the island of Bahrain, and in some of the coastal aquifers of the United Arab Emirates. A comparable situation has also arisen in the Nile Delta (Kotb, 2000), though here the cause is not necessarily solely groundwater overpumping, but may also be due to changes in water levels and freshwater recharge caused by the construction of the Aswan High Dam. Figure 4.5 shows the way in which chloride concentrations have increased and spread in the Llobregat Delta area of eastern Spain because of the incursion of seawater.

The Ghyben–Herzberg relationship (Figure 4.6) is based on the fact that freshwater has a lower density than saltwater, such that a column of seawater can support a column of freshwater approximately 2.5% higher than itself (or a ratio of about 40:41). So where a body of freshwater has accumulated in a reservoir rock or sediment which is also open to penetration from the sea, it does not simply lie flat on top of the saltwater but forms a lens, with a thickness that is approximately 41 times the elevation of the piezometric surface above sea level. The corollary of this rule is that if the hydrostatic pressure of the freshwater falls as a result of overpumping in a well, then the underlying saltwater will rise by 40 units for every unit by which the freshwater table is lowered. A rise in sea level can cause a comparably dramatic alteration in the balance between fresh and saltwater bodies. This is especially serious on low islands (Figure 4.7) (Broadus, 1990).

An example of concern about salinization following on from sea-level rise is the Shanghai area of China (Chen and Zong, 1999). The area is suffering subsidence as a result of delta sedimentation (at 3 mm per year) and groundwater extraction (now less than 10 mm

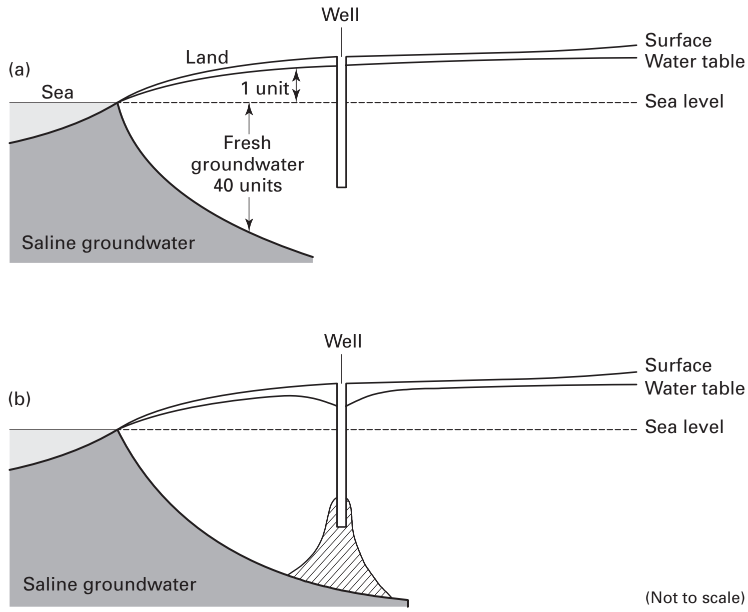

Figure 4.6 (a) The Ghyben–Herzberg relationship between fresh and saline groundwater. (b) The effect of excessive pumping from the well. The diagonal hatching represents the increasing incursion of saline water (after Goudie and Wilkinson, 1977, figure 63).

per year). It is very low-lying and in some counties the groundwater table is at 1.0–1.2 m. Thus a rise in sea level will raise the groundwater level, prolong waterlogging and cause a greater dominance of saline water in the deltaic area.

In any one location the causes of seawater incursion may be complex. A particularly good exemplification of this is provided by the coastal plains of northern Australia. Here Mulrennan and Woodroffe (1998) have assessed the potential role of such factors as sea-level change, rainfall variability, boat erosion of creeks, the activities of feral water buffalo, and sediment compaction.

Consequences of salinity

One consequence of the evaporative concentration of salts, and the pumping of saline waters back into rivers and irrigation canals from tubewells and other sources, is that river waters leading from irrigation areas show higher levels of dissolved salts. These, particularly when they contain nitrates, can make the water undesirable for human consumption.

A further problem is that, as irrigation water is concentrated by evapotranspiration, calcium and magnesium components tend to precipitate as carbonates, leaving sodium ions dominant in the soil solution. The sodium ions tend to be absorbed by colloidal clay particles, deflocculating them and leaving the resultant structureless soil almost impermeable to water and unfavorable to root development.

The death of vegetation in areas of saline patches, due both to poor soil structure and toxicity, creates bare ground that becomes a focal point for erosion by wind and water. Likewise, deflation from the desiccating surface of the Aral Sea causes large amounts of salt to be blown away in dust storms and to be deposited downwind. Some tens of millions of tonnes are being translocated by such means and their plumes are evident on satellite images.

Probably the most serious impact of salination is on plant growth. This takes place partly through its effect on soil structure, but more significantly through its effects on osmotic pressures and through direct toxicity. When a water solution containing large quantities of dissolved salts comes into contact with a plant cell it causes shrinkage of the protoplasmic lining. The phenomenon is due to the osmotic movement of the water, which passes from the cell towards the more concentrated soil solution. The cell collapses and the plant succumbs. Crop yields fall.

The toxicity effect varies with different plants and different salts. Sodium carbonate, by creating highly alkaline soil conditions, may damage plants by a direct caustic effect, whereas high nitrate may promote

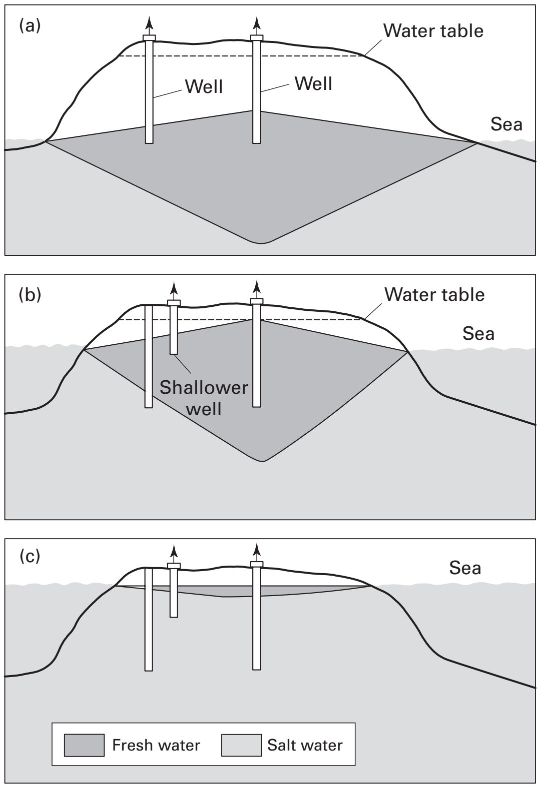

Figure 4.7 Impact of sea-level rise on an island water table. Note: the freshwater table extends below sea level 40 cm for every 1 cm by which it extends above sea level. (a) For islands with substantial elevation a 1-m rise in sea level simply shifts the entire water table up 1 m, and the only problem is that a few wells will have to be replaced with shallower wells (b). For very low islands, however, the water table cannot rise due to runoff, evaporation, and transpiration. A rise in sea level would thus narrow the water table by 40 cm for every 1 cm that the sea level rises (c), effectively eliminating groundwater supplies for the lowest islands (modified after Broadus, 1990).

undesirable vegetative growth in grapes or sugar beets at the expense of sugar content. Boron is injurious to many crop plants at solution concentrations of more than 1 or 2 ppm.

Anthropogenic increases in salinity are not new. Jacobsen and Adams (1958) have shown that they were a problem in Mesopotamian agriculture after about 2400 BC. Individual fields, which in 2400 BC were registered as salt-free, can be seen in the records of ancient temple surveyors to have developed conditions of sporadic salinity by 2100 BC. Further evidence is provided by crop choice, for the onset of salinization strongly favors the adoption of crops that are most salt-tolerant. Counts of grain impressions in excavated pottery from sites in southern Iraq dated at about 3500 BC suggest that at that time the proportions of wheat and barley were nearly equal. A little more than 1000 years later the less salt-tolerant wheat accounted for less than 20% of the crop, and by about 2100 BC it accounted for less than 2% of the crop. By 1700 BC the cultivation of wheat had been abandoned completely in the southern part of the alluvial plain. These changes in crop choice were accompanied by serious declines in yield, which can also probably be attributed to salinity. At 2400 BC the yield was 2537 L per hectare, by 2100 BC it was 1460, and by 1700 BC it was down to 897 L. It seems likely that this played an important part in the break-up of Sumerian civilization, although the evidence is not conclusive. Moreover, the Sumarians appear to have understood the problem and to have had coping strategies (Powell, 1985).

Reclamation of salt-affected lands

Because of the extent and seriousness of salinity, be the causes natural or anthropogenic, various reclamation techniques have been initiated. These can be divided into three main types: eradication, conversion, and control.

Eradication predominantly involves the removal of salt either by improved drainage or by the addition of quantities of freshwater to leach the salt out of the soil. Both solutions involve considerable expense and pose severe technological problems in areas of low relief and limited freshwater availability. Improved drainage can either be provided by open drains or by the use of tubewells (as at Mohenjo Daro, Pakistan) to reduce groundwater levels and associated salinity and waterlogging. A minor eradication measure, which may have some potential, is the biotic treatment of salinity through the harvesting of salt-accumulating plants such as *Suaeda fruticosa*.

Conversion involves the use of chemical methods to convert harmful salts into less harmful ones. For example, gypsum is frequently added to sodic soils to convert caustic alkali carbonates to soluble sodium sulfate and relatively harmless calcium carbonate:

$$Na_2CO_3 + CaSO_4 \leftrightarrow CaCO_3 + Na_2SO_4 \downarrow \text{leachable}$$

Some of the most effective ways of reducing the salinity hazard involve miscellaneous control measures, such as: less wasteful and lavish application of water through the use of sprinklers rather than traditional irrigation methods; the lining of the canals to reduce seepage; the realignment of canals through less permeable soil; and the use of more salt-tolerant plants. As salinity is a particularly serious threat at the time of germination and for seedlings, various strategies can be adopted during this critical phase of plant growth: plots can be irrigated lightly each day after seeding to prevent salt build-up; major leaching can be carried out just before planting; and areas to be seeded can be bedded in such a way that salts accumulate at the ridge tops, with the seed planted on the slope between the furrow bottom and the ridge top (Carter, 1975). Useful general reviews of methods for controlling soil salinity are given by Rhoades (1990) and Qadir et al. (2000).

Lateritization

In some parts of the tropics there are extensive sheets of a material called **laterite**, an iron and/or aluminum-rich duricrust (see Maignien, 1966, or Macfarlane, 1976). These iron-rich sheets result naturally, either because of a preferential removal of silica during the course of extensive weathering (leading to a *relative* accumulation of the sesquioxides of iron and aluminum), or because of an *absolute* accumulation of these compounds.

One of the properties of laterites is that they harden on exposure to air and through desiccation. Once hardened they are not favorable to plant growth. One particular way in which exposure may take place is by accelerated erosion, while forest removal may so cause a change in microclimate that desiccation of the laterite surface can take place. Indeed, one of the main problems with the removal of humid tropical rain forest is that soil hardening may occur. The phenomenon may occur in some, but by no means in all, so-called tropical soils (Richter and Babbar, 1991). Should hardening occur, it tends to limit the extent of successful soil utilization and severely retards the re-establishment of forest. Although Vine (1968: 90) and Sanchez and Buol (1975) have rightly warned against exaggerating this difficulty in agricultural land use, there are records from many parts of the tropics of accelerated induration brought about by forest removal (Goudie, 1973). In the Cameroons, for example, around 2 m of complete induration can take place in less than a century. In India, foresters have for a long time been worried by the role that plantations of teak (*Tectona grandis*) can play in lateritization. Teak is deciduous, demands light, likes to be well spaced (to avoid crown friction), dislikes competition from undergrowth, and is shallow-rooted. These characteristics mean that teak plantations tend to expose the soil surface to erosive and desiccative forces more than does the native vegetation cover.

One of the main exponents of the role that human agency has played in lateritization in the tropical world has been Gourou (1961: 21–2). Although he may be guilty of exaggerating the extent and significance of laterite, Gourou gives many examples from low latitudes of falling agricultural productivity resulting from the onset of lateritization. It is worth quoting him at length:

> On the whole, laterite is hostile to agriculture owing to its sterility and compactness. All tropical countries have not reached the same degree of lateritic 'suicide' but when the evolution has advanced a considerable way, man is placed in very strange conditions . . . Laterite is a pedological leprosy. Man's activities aggravate the dangers of laterite and increase the rate of the process of lateritization. To begin with, erosion when started by negligent removal of the forest simply wears away the friable and relatively fertile soil which would otherwise cover the laterite and support forest or crops . . . The forest checks the formation of the laterite in various ways. The trees supply plenty of organic matter and maintain a good proportion of humus in the soil. The action of capillary attraction is checked by the loosening of the soil; and the bases are retained through the absorbent capacity of humus. The forest slows down evaporation from the soil . . . it reduces percolation and consequently leaching. Lastly, the forest may improve the composition of the soil by fixing atmospheric dust.

Accelerated podzolization and acidification

There is an increasing amount of evidence that the introduction of agriculture, deforestation, and pastoralism to parts of upland western Europe promoted some major changes in soil character: notably an increase in the development of acidic and **podzolized** conditions, associated with the development of peat bogs. Climatic changes may have played a role, as could progressive leaching of last glacial drifts during the passage of the Holocene. But the association in time and space of human activities with soil deterioration has become increasingly clear (Evans et al., 1975).

Replacing the natural forest vegetation with cultivation and pasture, human societies set in train various related processes, especially on base-poor materials. First, the destruction of deep-rooting trees curtailed the enrichment of the surface of the soil by bases brought up from the deeper layers. Second, the use of fire to clear forest may have released nutrients in the form of readily soluble salts, some of which were inevitably lost in drainage, especially in soils poor in colloids (Dimbleby, 1974). Third, the taking of crops and animal products depleted the soil reserves to an extent probably greater than that arising from any of the manuring practices of prehistoric settlements. Fourth, as the soil degraded, the vegetation which invaded – especially bracken and heather – itself tended to produce a more acidic humus type of soil than the original mixed deciduous forest, and so continued the process.

Various workers now attribute much of the podzolization in upland Britain to such processes, and Dimbleby has concluded that 'although a few soils have been podzols since the Atlantic period [in the mid-Holocene], the majority are secondary, having arisen as a result of man's assault on the landscape, particularly in the Bronze Age' (see also Bridges, 1978).

The development of podzols and their associated ironpans (Cunningham et al., 2001), by impeding downward percolating waters, may have accelerated the formation of peats, which tend to develop where there is waterlogging through impeded drainage. Many peat bogs in highland Britain appear to coincide broadly in age with the first major land-clearance episodes (Merryfield and Moore, 1971; Moore, 1973). Another fact that would have contributed to their development is that when a forest canopy is removed (as by deforestation) the transpiration demand of the vegetation is reduced, less rainfall is intercepted, so that the supply of groundwater is increased, aggravating any waterlogging.

However, the role of natural processes must not be totally forgotten, and Ball's assessment would seem judicious (1975: 26):

> It seems to be on balance that the highland trends in soil formation due to climate, geology and relief have been clearly running in the direction of leaching, acidity, podzolization, **gleying** and peat formation. For the British highlands generally, man has only intervened to hasten or slow the rate of these trends, rather than being in a position to alter the whole trend from one pedogenetic trend to another.

Moreover, it would be plainly misleading to stress only the deleterious effects of human actions on European soils. Traditional agricultural systems have often employed laborious techniques to augment soil fertility and to reduce such properties as undesirable acidity. In Britain, for example, the addition of chalk to light sandy land goes back at least to Roman times and the marl pits from which the chalk was dug are a striking feature of the Norfolk landscape, where Prince (1962) has identified at least 27,000 hollows. Similarly, in The Netherlands, Germany, and Belgium there are soils which for centuries (certainly more than 1000 years) have been built up (often over 50 cm) and fertilized with a mixture of manure, sods, litter, or sand. Such soils are called Plaggen soils (Pope, 1970). Plaggen soils also occur in Ireland, where the addition of sea-sand to peat was carried out in pre-Christian times. Likewise, before European settlement in New Zealand, the Maoris used thousands of tonnes of gravel and sand, carried in flax baskets, to improve soil structure (Cumberland, 1961).

Another type of soil that owes much to human influence is the category called 'paddy soil'. Long-continued irrigation, leveling, and manuring of terraced land in China and elsewhere have changed the nature of the pre-existing soils in the area. Among the most important modifications that have been recognized (Gong, 1983; Zhang and Gong, 2003) are an increase in organic matter, and increase in base saturation, and the translocation and reduction of iron and manganese.

One serious type of soil acidification is that which produces acid sulfate soils. As Dent and Pons (1995: 263) wrote:

> Acid sulphate soils are the nastiest soils in the world. They generate sulphuric acid that brings their pH as low as 2 and leaks into drainage and floodwaters. In this acid environment, aluminium and other toxic elements kill vegetation and aquatic life or, in sub-lethal doses, render many species stunted and sickly. Generations of people depending on these soils have been impoverished and, probably, poisoned by their drinking water.

The reason for the development of such extremely acid soils is that originally they accumulated as sediments under severely reducing conditions in environments such as tidal (e.g., mangrove) swamps, or brackish lakes. Large amounts of sulfitic mud accumulated. When such materials are drained reduction is replaced by oxidation, and sulfuric acid is produced. Infamous examples are known from the drained polders of The Netherlands and from the drained coastal swamps of southeast Asia.

Some soils are currently being acidified by air pollution and the deposition of acid precipitation (Grieve, 2001) (see also Chapter 7). Many soils have a resistance to acidification because of their buffering capacity, which enables them to neutralize acidity. However, this resistance very much depends on soil type and situation, and soils which have a low buffering capacity because of their low calcium content (as, e.g., on granite), and which are subjected to high levels of precipitation, may build up high levels of acidity.

The concept of **critical loads** has been developed. They are defined as exposures below which significant harmful effects on sensitive elements of the environment do not occur according to currently available knowledge. The critical load for sensitive forest soils on gneiss, granite or other slow-weathering rocks is often less than 3 kg of sulfur per hectare per year. In some of the more polluted parts of central Europe, the rates of sulfur deposition may be between 20 and 100 kg per hectare per year (Ågren and Elvingson, 1996).

The immediate impact of high levels of acid input to soils is to increase the exchange between hydrogen (H^+) ions and the nutrient cations, such as potassium (K^+), magnesium (Mg^{2+}), and calcium (Ca^{2+}). As a result of this exchange, such cations can be quickly leached from the soil, along with the sulfate from the acid input. This leaching leads to nutrient deficiency. Acidification also leads to a change in the rate at which dead organic matter is broken down by soil microbes. It can also render some ions, such as aluminum, more mobile and this has been implicated in the phenomenon of forest decline (see Chapter 2).

Soil carbon

One important consequence of land use and land cover changes is changes in the carbon content of soils. This has potential significance in terms of carbon release to the atmosphere and hence for the enhanced greenhouse effect (Lal et al., 1995). Soil organic carbon includes plant, animal and microbial residues.

Under prolonged cultivation there is strong evidence for loss of soil carbon. Agricultural practices that contribute to this are deforestation and biomass burning, drainage of wetlands, plowing and other forms of soil disturbance, and removal of crop residues. Soil carbon is depleted by oxidation or mineralization due to breakdown of aggregates, leading to exposure of carbon, leaching, and translocation of dissolved organic carbon or particulate organic carbon, and accelerated erosion by runoff or wind (Lal, 2002). Losses of soil organic carbon of as much as 50% in surface soils (20 cm) have been observed after cultivation for 30–50 years. Reductions average around 30% of the original amount in the top 100 cm (Post and Kwon, 2000).

Much depends, however, on the nature of land management. In recent decades, for example, the carbon stock in agricultural land in the USA may have increased because of the adoption of conservation tillage practices on cropland and a reduction in the use of bare fallow (Eve et al., 2002). The use of nitrogenous fertilizers may also explain some of the increase (Buyanovsky and Wagner, 1998).

Conversely, reafforestation and conversion of cropland to grassland can cause substantial gains in soil organic carbon. In general, after afforestation there may be a period when soil organic carbon declines, because of low rates of litter fall and continuing decomposition of residues from the preceding agricultural phase. As the forest cover develops, inputs of carbon exceed outputs. The rates of building of carbon vary, and tend to be greater in cool and humid regions and under hardwoods and softwoods rather than under *Eucalyptus* or *Pinus radiata* (Paul et al., 2002).

Soil structure alteration

One of the most important features of a soil, in terms of both its suitability for plant growth and its inherent erodibility, is its structure. There are many ways in which humans can alter this, especially by compacting it with agricultural machinery (Horn et al., 2000), by the use of recreation vehicles, by changing its chemical character through irrigation, and by trampling

Table 4.7 Change to soil properties resulting from the passage of 100 motorcycles in New Zealand. Source: Crozier et al. (1978, table 1)

Soil property	*Total number of sites*	*Number of sites with significant change**	*Mean percentage at significant sites*	*Number of sites with significant increase*	*Number of sites with significant decrease*	*Main direction of change*	*Mean percentage change in direction*
Infiltration capacity	16	16 (100%)	84.3	3	13	Decrease	78.1
Bearing capacity	21	10 (48%)	22.8	2	8	Decrease	18.6
Soil moisture	20	14 (70%)	15.5	5	9	Decrease	16.7
Dry bulk density	19	11 (58%)	13.6	9	2	Decrease	13.3

*Change is significant when greater than: 10% for infiltration capacity; 10% for bearing capacity; 5% for soil moisture; 5% for bulk density.

(Grieve, 2001). Soil compaction, which involves the compression of a mass of soil into a smaller volume, tends to increase the resistance of soil to penetration by roots and emerging seedlings, and limits oxygen and carbon dioxide exchange between the root zone and the atmosphere. Moreover, it reduces the rate of water infiltration into the soil, which may change the soil moisture status and accelerate surface runoff and soil erosion (Chancellor, 1977). For example, the effects of the passage of vehicles on some soil structural properties are shown in Table 4.7. Excessive use of heavy agricultural machinery is perhaps the major cause of soil compaction, and most procedures in the cropping cycle, from tillage and seedbed preparation, through drilling, weeding and agrochemical application to harvesting, are now largely mechanized, particularly in the developed world. Most notable of all is the reduction that is caused in soil **infiltration capacity**, which may explain why vehicle movements can often lead to gully development. Whether one is dealing with primitive sledges (as in Swaziland), or with the latest recreational toys of leisured Californian adolescents, the effects may be comparable.

Grazing is another activity that can damage soil structure through trampling and compaction. Heavily grazed lands tend to have considerably lower infiltration capacities than those found in ungrazed lands. Trimble and Mendel (1995) have drawn particular attention to the capability that cows have to cause soil compaction. Given their large mass, their small hoof area, and the stress that may be imposed on the ground when they are scrambling up a slope, they are probably remarkably effective in compacting soils. The removal of vegetation cover and associated litter also changes infiltration capacity, since cover protects the soil from packing by raindrops and provides organic matter for binding soil particles together in open aggregates. Soil fauna that live on the organic matter assist this process by churning together the organic material and mineral particles. Dunne and Leopold have ranked the relative influence of different land-use types on infiltration (1978, table 6.2, after US Soil Conservation Service):

Highest infiltration	Woods, good
	Meadows
	Woods, fair
	Pasture, good
	Woods, poor
	Pasture, fair
	Small grains, good rotation
	Small grains, poor rotation
	Legumes after row crops
	Pasture, poor
	Row crops, good rotation (more than one quarter in hay or sod)
	Row crops, poor rotation (one quarter or less in hay or sod)
Lowest infiltration	Fallow

In general, experiments show that reafforestation improves soil structure, especially the pore volume of the soils (see, e.g., Challinor, 1968). Plowing is also known to produce a compacted layer at the base of the zone of plowing (Baver et al., 1972). This layer has been termed the ‘plow sole’. The normal action of the plow is to leave behind a loose surface layer and a dense subsoil where the soil aggregates have been pressed together by the sole of the plow. The compacting action can be especially injurious when the depth of plowing is both constant and long term, and when heavy machinery is used on wet ground (Greenland and Lal, 1977).

On the other hand, for many centuries farmers have achieved improvements in soil structure by deliberate practice, particularly with a view to developing the all-important crumb structure. In pre-Roman times people in Britain and France added lime to heavy clay soils, while the agricultural improvers of the eighteenth and nineteenth centuries improved the structure of sandy heath soils by adding clay, and of clay soils by adding calcium carbonate (marl). They also compacted such soils and added binding organic matter by breeding sheep and feeding them with turnips and other fodder plants (Russell, 1961).

In the modern era attempts have been made to reduce soil crusting by applying municipal and animal wastes to farm land, by adding chemicals such as phosphoric acid, and by adopting a cultivation system of the no-tillage type. The last practice is based on the idea that the use of herbicides has eliminated much of the need for tillage and cultivation in row crops; seeds are planted directly into the soil without plowing, and weeds are controlled by the herbicides. With this method less bare soil is exposed and heavy farm machinery is less likely to create soil compaction problems (see Carlson, 1978, for some of these methods).

Soil structures may also be modified to increase water runoff, particularly in arid zones where the runoff obtained can augment the meager water supply for crops, livestock, industrial and urban reservoirs, and groundwater recharge projects. In the Negev farming was practiced in this way, especially in the Nabatean and the Romano-Byzantine periods (about 300 BC to AD 630), and attempts were made to induce runoff by clearing the surface gravel of the soil and heaping it into thousands of mounds. This exposed the finer silty soil beneath, facilitating soil crust formation by raindrop impact, decreasing infiltration capacity and reducing surface roughness, so that runoff increased (Evenari et al., 1971). Today a greater range of techniques are available for the same end: soils can be smoothed and compacted by heavy machinery, soil crusting can be promoted by dispersion of soil colloids with sodium salts, the permeability of the soil surface can be reduced by applying water-repellent materials, and soil pores can be filled with binders (Hillel, 1971).

Soil drainage and its impact

Soil drainage ‘has been a gradual process and the environmental changes to which it has led have, by reason of that gradualness, often passed unnoticed’ (Green, 1978: 171). To be sure, the most spectacular feats of drainage – arterial drainage – involving the construction of veritable rivers and large dike systems, as seen in The Netherlands and the Fenlands of eastern England, have received attention. However, more widespread than arterial drainage, and sometimes independent of it, is the drainage of individual fields, either by surface ditching or by underdrainage with tile pipes and the like. Green (1978) has attempted to map the areas of drained agricultural land. In Finland, Denmark, Great Britain, The Netherlands, and Hungary, the majority of agricultural land is drained (Figure 4.8).

In Britain underdrainage was promoted by government grants and reached a peak of about one million hectares per year in the 1970s in England and Wales. More recently government subsidies have been cut and the uncertain economic future of farming has led to a reduction in farm expenditure. Both tendencies have led to a reduction in the rate of field drainage, which by 1900 was only 40,000 hectares per year (Robinson, 1990).

The drainage conditions of the soil have also frequently been altered by the development of ridge and furrow patterns created by plowing. Such patterns are a characteristic feature of many of the heavy soils of lowland England where large areas, especially of the Midland lowlands, are striped by long, narrow ridges of soil, lying more or less parallel to each other and usually arranged in blocks of approximately rectangular shape. They were formed by plowing with heavy plows pulled by teams of oxen; the precise mechanism has been described with clarity by Coones and Patten (1986: 154).

Soil drainage has been one of the most successful ways in which communities have striven to increase agricultural productivity; it was practiced by Etruscans, Greeks and Romans (Smith, 1976). Large areas of marshland and floodplain have been drained to human advantage. By leading water away, the water table is lowered and stabilized, providing greater depth for the root zone. Moreover, well-drained soils warm up

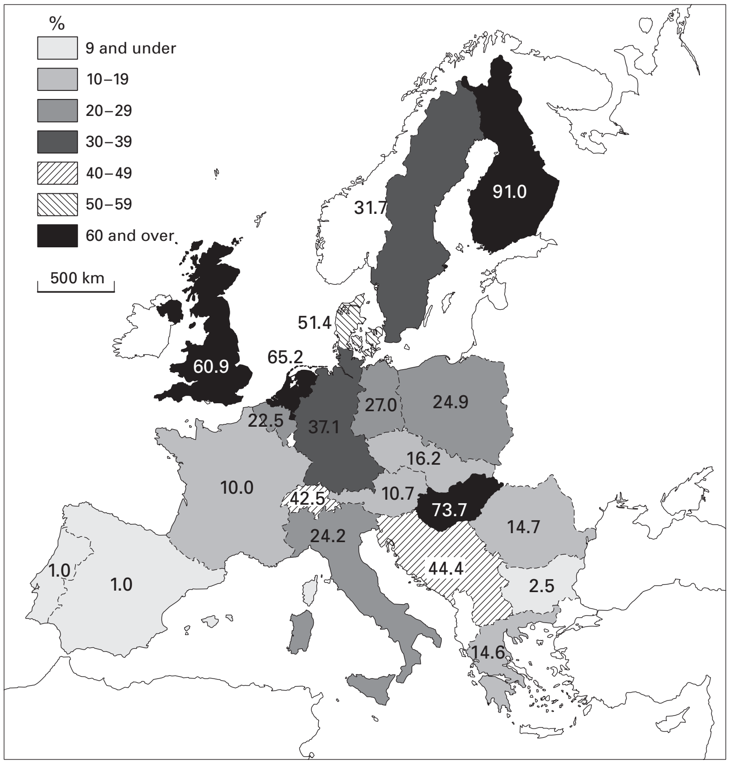

Figure 4.8 Percentage of drained agricultural land in Europe. There are no data for the blank areas (after Green, 1978, figure 1).

earlier in the spring and thus permit earlier planting and germination of crops. Farming is easier if the soil is not too wet, since the damage to crops by winter freezing may be minimized, undesirable salts carried away, and the general physical condition of the soil improved. In addition, drained land may have certain inherent virtues: tending to be flat it is less prone to erosion and more amenable to mechanical cultivation. It will also be less prone to drought risk than certain other types of land (Karnes, 1971). Paradoxically, by reducing the area of saturated ground, drainage can alleviate flood risk in some situations by limiting the extent of a drainage basin that generates saturation excess overland flow.

Conversely, soil drainage can have quite undesirable or unplanned effects. For example, some drainage systems, by raising drainage densities, can increase flood risk by reducing the distance over which unconcentrated overland flow (which is relatively slow) has to travel before reaching a channel (where flow is relatively fast). In central Wales, for instance, the establishment of drainage ditches in peaty areas to enable afforestation has tended to increase flood peaks in the rivers Wye and Severn (Howe et al., 1966).

Drainage can also cause long-term damage to soil quality. A fall in water level in organic soils can lead to the oxidation and eventual disappearance of peaty materials, which in the early stages of post-drainage use may be highly productive. This has occurred in the English Fenland, and in the Everglades of Florida, where drainage of peat soils has led to a subsidence in the soil of 32 mm per year (Stephens, 1956).

The soil climate of neighboring areas may also be modified when the water tables of drained land are lowered. This has been known to create problems for forestry in areas of marginal water availability.

Soil moisture content can also determine the degree to which soils are subjected to expansion and contraction effects, which in turn may affect engineering structures in areas with expansive soils (Holtz, 1983). Soils containing sodium montmorillonite type clays, when drained or planted with large trees, may dry out and cause foundation problems (Driscoll, 1983).

Some of the most contentious effects of drainage are those associated with the reduction of wetland wildlife habitats. In Britain this has become an important political issue, especially in the context of the Somerset Levels and the Halvergate Marshes.

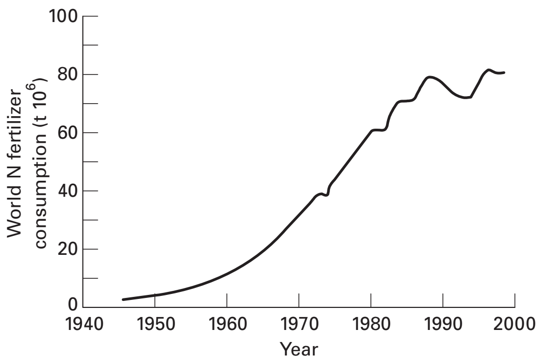

Figure 4.9 Global consumption of nitrogen fertilizer over the past five decades.

Soil fertilization

The chemistry of soils has been changed deliberately by the introduction of chemical fertilizers. The employment of chemical fertilizers on a large scale is little more than 150 years old. In the early nineteenth century nitrates were first imported from Chile, and sulfate of ammonia was produced only after the 1820s as a by-product of coal gas manufacture. In 1843 the first fertilizer factory was established at Deptford Creek (London), but for a long time superphosphates were the only manufactured fertilizers in use. In the twentieth century synthetic fertilizers, particularly nitrates, were developed, notably by Scandinavian countries that used their vast resources of waterpower. Potassic fertilizers came into use much later than the phosphatic and nitrogenous; nineteenth-century farmers hardly knew them (Russell, 1961). The rise in nitrogen fertilizer consumption for the world as a whole between 1950 and 1990 is shown in Figure 4.9. Global fertilizer consumption in 2000 was around 141 million tonnes of nutrient, of which around 61% was nitrogen, 23% phosphate, and 16% potash. The use of synthetic fertilizers has greatly increased agricultural productivity in many parts of the world, and remarkable increases in yields have been achieved. It is also true that in some circumstances proper fertilizer use can help minimize erosion by ensuring an ample supply of roots and plant residues, particularly on infertile or partially degraded soils (Bockman et al., 1990).

On the other hand, the increasing use of synthetic fertilizers can create environmental problems such as water pollution, while their substitution for more traditional fertilizers may accelerate soil-structure deterioration and soil erosion. One effect has been the increase in the water repellency of some surface soil materials. This in turn can reduce soil infiltration rates and a consequential increase in erosion by overland flow (Conacher and Conacher, 1995: 39). Fertilizers can also promote soil acidity and may lead to deficiencies or toxic excesses of major nutrients and trace elements. They may also contain impurities, such as fluoride, lead, cadmium, zinc, and uranium. Some of these heavy metals can inhibit water uptake and plant growth. They may also become concentrated in food crops, which can have important implications for human health.

Fires and soil quality

The importance and antiquity of fire as an agency through which the environment is transformed requires that some attention be given to the effects of fire on soil characteristics.

Fire has often been used intentionally to change soil properties, and both the release of nutrients by fire and the value of ash have long been recognized, notably by those involved in shifting agriculture based on slash-and-burn techniques. Following cultivation, the loss of nutrients by leaching and erosion is very rapid (Nye and Greenland, 1964), and this is why after only a few years the shifting cultivators have to move on to new plots. Fire rapidly alters the amount, form, and distribution of plant nutrients in ecosystems, and, compared with normal biological decay of plant remains, burning rapidly releases some nutrients into a plant-available form. Indeed, the amounts of P, Mg, K, and Ca released by burning forest and scrub vegetation are high in relation to both the total and available quantities of these elements in soils (Raison, 1979). In particular phosphate loss can be very detrimental to soil fertility (Thomas et al., 2000). In forests, burning often causes the pH of the soil to rise by three units or more, creating alkaline conditions where formerly there was acidity. Burning also leads to some direct nutrient loss by volatilization and convective transfer of ash, or by loss of ash to water erosion or wind deflation. The removal of the forest causes soil temperatures to increase because of the absence of shade, so that humus is often lost at a faster rate than it is formed (Grigg, 1970).

Table 4.8 The beneficial properties of humus. Source: Swift and Sanchez (1984, table 2)

Property		*Explanation*	*Effect*
Chemical	Mineralization	Decomposition of humus yields CO_2, NH_4^+, NO_3^-, PO_4^{3-}, and SO_4^{2-}	A source of nutrient elements for plant growth
	Cation exchange	Humus has negatively charged surfaces which bind cations such as Ca^+ and K^+	Improved cation exchange capacity (CEC) of soil. From 20 to 70% of the CEC of some soils (e.g., Mollisols) is attributable to humus
	Buffer action	Humus exhibits buffering in slightly acid, neutral and alkaline ranges	Helps to maintain a uniform pH in the soil
	Acts as a matrix for biochemical action in soil	Binds other organic molecules electrostatically or by covalent bonds	Affects bioactivity persistence and biodegradability of pesticides
	Chelation	Forms stable complexes with Cu^{2+}, Mn^{2+}, Zn^{2+}, and other polyvalent cations	May enhance the availability of micronutrients to higher plants
Physical	Water retention	Organic matter can hold up to 20 times its weight in water	Helps prevent drying and shrinking. May significantly improve moisture-retaining properties of sandy soils
	Combination with clay	Cements soil particles into structural units called aggregates	Improves aeration. Stabilizes structure. Increases permeability
	Color	The typical dark color of many soils is caused by organic matter	May facilitate warming

Soil erosion: general considerations

Loss of soil humus, whether as a result of fire, drainage, deforestation or plowing, is an especially serious manifestation of human alteration of soil. As Table 4.8 indicates, humus has many beneficial effects on both the chemical and the physical properties of soil. Its removal by human activity can be a potent contributory cause of soil erosion.

The scale of accelerated soil erosion that has been achieved by human activities has been well summarized by Myers (1988: 6):

> Since the development of agriculture some 12,000 years ago, soil erosion is said by some to have ruined 4.3 million km^2 of agricultural lands, or an area equivalent to rather more than one-third of today's crop-lands . . . the amount of agricultural land now being lost through soil erosion, in conjunction with other forms of degradation, can already be put at a minimum of 200,000 km^2 per year.

That soil erosion is a major and serious aspect of the human role in environmental change is not to be doubted (Sauer, 1938). There is a long history of weighty books and papers on the subject (see, e.g., Marsh, 1864; Bennett, 1938; Jacks and Whyte, 1939; Morgan, 1995). Although many techniques have been developed to reduce the intensity of the problem (see Hudson, 1987) it appears to remain intractable. As L. J. Carter (1977: 409) has reported of the USA:

> Although nearly $15 billion has been spent on soil conservation since the mid-1930s, the erosion of croplands by wind and water . . . remains one of the biggest, most pervasive environmental problems the nation faces. The problem's surprising persistence apparently can be attributed at least in part to the fact that, in the calculation of many farmers, the hope of maximizing short-term crop yields and profits has taken precedence over the longer term advantages of conserving the soil. For even where the loss of topsoil has begun to reduce the land's natural fertility and productivity, the effect is often masked by the positive response to heavy application of fertilizer and pesticides, which keep crop yields relatively high.

Although construction, urbanization, war, mining, and other such activities are often significant in accelerating the problem, the prime causes of soil erosion are deforestation and agriculture (Figure 4.10). Pimentel (1976) estimated that in the USA soil erosion on agricultural land operates at an average rate of about 30 tonnes per hectare per year, which is approximately eight times quicker than topsoil is formed. He calculated that water runoff delivers around 4 billion tonnes

Figure 4.10 Soil erosion near Baringo in Kenya has exposed the roots of a tree, thereby indicating the speed at which soil can be lost.

of soil each year to the rivers of the 48 contiguous states, and that three-quarters of this comes from agricultural land. He estimated that another billion tonnes of soil is eroded by the wind, a process that created the Dust Bowl of the 1930s. More recently, Pimentel et al. (1995) have argued that in the USA about 90% of cropland is losing soil above the sustainable rate, that about 54% of pasture land is overgrazed and subject to high rates of erosion, and that erosion costs about $44 billion each year. They argue that on a global basis soil erosion costs the world about $400 billion each year. However, as Trimble and Crosson (2000) and Boardman (1998) point out, determination of general rates of soil erosion is fraught with uncertainties.

One serious consequence of accelerated erosion is the sedimentation that takes place in reservoirs, shortening their lives and reducing their capacity. Many small reservoirs, especially in semi-arid areas and in areas with erodible sediments in their catchments such as the loess lands of China, appear to have an expected life of only 30 years or even less (see, e.g., Rapp et al., 1972). Soil erosion also has serious implications for soil productivity. A reduction in soil thickness reduces available water capacity and the depth through which root development can occur. The water-holding properties of the soil may be lessened as a result of the preferential removal of organic material and fine sediment. Hardpans and duricrusts may become exposed at the surface, and provide a barrier to root penetration. Furthermore, splash erosion may cause soil compaction and crusting, both of which may be unfavorable to germination and seedling establishment. Erosion also removes nutrients preferentially from the soil. Some damage may be caused by associated excessive sedimentation, while wind erosion may lead to the direct sandblasting of crops. Finally, extreme erosion may lead to wholesale removal of both seeds and fertilizer. Stocking (1984) provides a useful review of these problems.

Soil erosion associated with deforestation and agriculture

Forests protect the underlying soil from the direct effects of rainfall, generating what is generally an environment in which erosion rates tend to be low. The canopy plays an important role by shortening the fall of raindrops, decreasing their velocity and thus reducing kinetic energy. There are some examples of certain types (e.g., beech) in certain environments (e.g., maritime temperate) creating large raindrops, but in general most canopies reduce the erosion effects of rainfalls. Possibly more important than the canopy in reducing erosion rates in forest is the presence of humus in forest soils (Trimble, 1988). This both absorbs the impact of raindrops and has an extremely high permeability. Thus forest soils have high infiltration capacities. Another reason that forest soils have an ability to transmit large quantities of water through their fabrics is that they have many macropores produced by roots and their rich soil fauna. Forest soils are also well aggregated, making them resistant to both wetting and water drop impact. This superior degree of aggradation is a result of the presence of considerable organic material, which is an important cementing agent in the formation of large water-stable aggregates. Furthermore, earthworms also help to produce large aggregates. Finally, deep-rooted trees help to stabilize steep slopes by increasing the total shear strength of the soils.

It is therefore to be expected that with the removal of forest, for agriculture or for other reasons, rates of soil loss will rise (Figure 4.11) and mass movements will increase in magnitude and frequency. The rates of erosion that result will be particularly high if the ground is left bare; under crops the increase will be less marked. Furthermore, the method of plowing, the

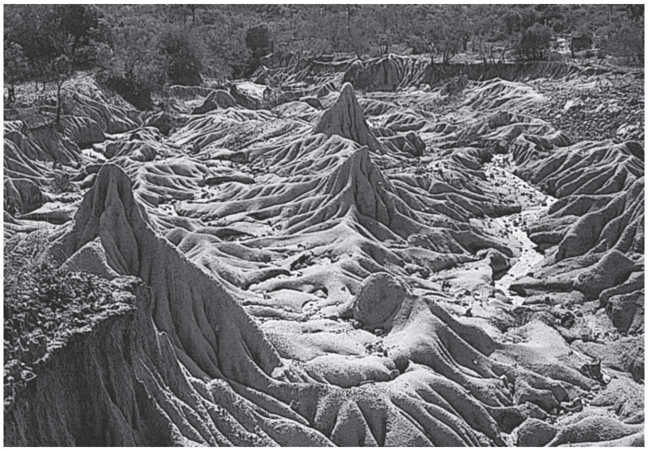

Figure 4.11 The removal of vegetation in Swaziland creates spectacular gully systems, which in southern Africa are called *dongas*. The smelting of local iron ores in the early nineteenth century required the use of a great deal of firewood, which may have contributed to the formation of this example.

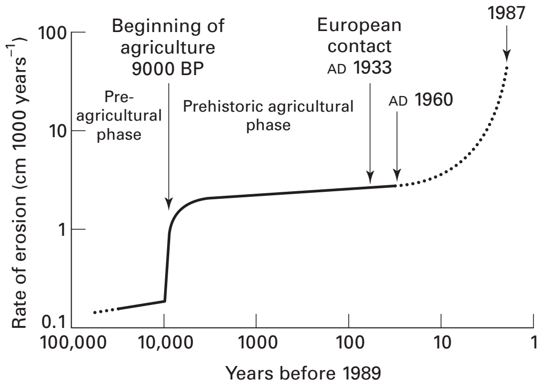

Figure 4.12 Rates of erosion in Papua New Guinea in the Holocene derived from rates of sedimentation in Kuk Swamp (after Hughes et al., 1991, figure 5, with modifications).

time of planting, the nature of the crop, and size of the fields, will all have an influence on the severity of erosion.

It is seldom that we have reliable records of rates of erosion over a sufficiently long time-span to show just how much human activities have accelerated these effects, and it is important to try and isolate the role of human impacts from climatic changes (Wilby et al., 1997). Recently, however, techniques have been developed which enable rates of erosion on slopes to be gauged over a lengthy time-span by means of dendrochronological techniques that date the time of root exposure for suitable species of tree. In Colorado, USA, Carrara and Carroll (1979) found that rates over the past 100 years have been about 1.8 mm per year, whereas in the previous 300 years rates were between 0.2 and 0.5 mm per year, indicating an acceleration of about sixfold. This great jump has been attributed to the introduction of large numbers of cattle to the area about a century ago.

Another way of obtaining long-term rates of soil erosion is to look at rates of sedimentation on continental shelves and on lake floors. The former method was employed by Milliman et al. (1987) to evaluate sediment removal down the Yellow River in China during the Holocene. They found that, because of accelerated erosion, rates of sediment accumulation on the shelf over the past 2300 years have been ten times higher than those for the rest of the Holocene (i.e., since around 10,000 years BP).

Another good example of using long-term sedimentation rates to infer long-term rates of erosion is provided by Hughes et al.'s (1991) study of the Kuk Swamp in Papua New Guinea (Figure 4.12). They identify low rates of erosion until 9000 years BP, when, with the onset of the first phase of forest clearance, erosion rates increased from 0.15 cm per 1000 years to about 1.2 cm per 1000 years. Rates remained relatively stable until the past few decades when, following European contact, the extension of anthropogenic grasslands, subsistence gardens, and coffee plantations has produced a rate that is very markedly higher: 34 cm per 1000 years.

A further good long-term study of the response rates of erosion to land use changes is provided by a study undertaken on the North Island of New Zealand by Page and Trustrum (1997). During the past 2000 years of human settlement their catchment has undergone a change from indigenous forest to fern/scrub following Polynesian settlement (*c.* 560 years BP) and then a change to pasture following European settlement (AD 1878). Sedimentation rates under European pastoral land use are between five and six times the rates that occurred under fern/scrub and between eight and seventeen times the rate under indigenous forest. In a broadly comparable study, Sheffield et al. (1995) looked at rates of infilling of an estuary fed by a sheepland catchment in another part of New Zealand. In pre-Polynesian times rates of sedimentation were 0.1 mm per year, during Polynesian times the rate

Table 4.9 Runoff and erosion under various covers of vegetation in parts of Africa. Source: After Charreau (table 5.5, p. 153) in Greenland and Lal (1977)

Locality	*Average annual rainfall (mm)*	*Slope (%)*	*Annual runoff* (%)*			*Erosion* (t ha^{-1} year^{-1})*		
			A	*B*	*C*	*A*	*B*	*C*
Ouagadougou (Burkina Faso)	850	0.5	2.5	2–32	40–60	0.1	0.6–0.8	10–20
Sefa (Senegal)	1300	1.2	1.0	21.2	39.5	0.2	7.3	21.3
Bouake (Ivory Coast)	1200	4.0	0.3	0.1–26	15–30	0.1	1–26	18–30
Abidjan (Ivory Coast)	2100	7.0	0.1	0.5–20	38	0.03	0.1–90	108–170
Mpwapwa† (Tanzania)	*c.* 570	6.0	0.4	26.0	50.4	0	78	146

*A = forest or ungrazed thicket; B = crop; C = barren soil.
†From Rapp et al. (1972: 259, figure 5).

climbed to 0.3 mm per year, and since European land clearance in the 1880s the rate has shot up to 11 mm per year (see also Nichol et al., 2000). A good case study of the effect of European settlement on soil erosion rates in neighboring Australia is given by Olley and Wasson (2003).

Rates of sediment accumulation on floodplains also give an indication of historical rates of soil erosion. This is a topic that has been well reviewed by Knox (2002) and which is discussed further in Chapter 6.

In a more general sense there are plainly huge difficulties in estimating erosion rates in pre-human times, but in a recent analysis McLennan (1993) has estimated that the pre-human suspended sediment discharge from the continents was about 12.6×10^{15} grams per year, which is about 60% of the present figure.

Table 4.9, which is based on data from tropical Africa, shows the comparative rates of erosion for three main types of land use: trees, crops, and barren soil. It is very evident from these data that under crops, but more especially when ground is left bare or under fallow, soil erosion rates are greatly magnified. At the same time, and causally related, the percentage of rainfall that becomes runoff is increased.

In some cases the erosion produced by forest removal will be in the form of widespread surface stripping. In other cases the erosion will occur as more spectacular forms of mass movement, such as mudflows, landslides, and debris avalanches. Some detailed data on debris-avalanche production in North American catchments as a result of deforestation and forest road construction are presented in Table 4.10. They illustrate the substantial effects created by clear-cutting and by the construction of logging roads. It is indeed probable that a large proportion of the erosion associated with forestry operations is caused by road construction, and care needs to be exercised to minimize these effects. The digging of drainage ditches in upland pastures and peat moors to permit tree-planting in central Wales has also been found to cause accelerated erosion (Clarke and McCulloch, 1979), while the elevated sediment loads can cause reservoir pollution (Burt et al., 1983).

In general, the greater the deforested proportion of a river basin the higher the sediment yield per unit area will be. In the USA the rate of sediment yield appears to double for every 20% loss in forest cover.

Soil erosion resulting from deforestation and agricultural practice is often thought to be especially serious in tropical areas or semi-arid areas (see Moore, 1979, for a good case study), but it is also a problem in the UK (Figure 4.13), in mainland Europe (Fuller, 2003), and in Russia (Sidorchuk and Golosov, 2003). Measurements by Morgan (1977) on sandy soils in the English East Midlands near Bedford indicate that rates of soil loss under bare soil on steep slopes can reach 17.69 tonnes per hectare per year, compared with 2.39 under grass and nothing under woodland (Table 4.11), and subsequent studies have demonstrated that water-induced soil erosion is a substantial problem, in spite of the relatively low erosivity of British rainfall. Walling and Quine (1991: 123) have identified the following farming practices as contributing to this developing problem.

1 Plowing up of steep slopes that were formerly under grass, in order to increase the area of arable cultivation.

Table 4.10 Debris-avalanche erosion in forest, clear-cut, and roaded areas. Source: after Swanston and Swanson, 1976, table 4

Site	Area type	Period of records (years)	Area (%)	Area (km^2)	Number of slides	Debris-avalanche erosion ($m^3\ km^{-2}\ yr^{-1}$)	Rate of debris-avalanche erosion relative to forested areas
Stequaleho Creek, Olympic Peninsula	Forest	84	79	19.3	25	71.8	× 1.0
	Clear-cut	6	18	4.4	0	0	0
	Road	6	3	0.7	83	11,825	× 165
	Total			24.4	108		
Alder Creek, western Cascade Range, Oregon	Forest	25	70.5	12.3	7	45.3	× 1.0
	Clear-cut	15	26.0	4.5	18	117.1	× 2.6
	Road	15	3.5	0.6	75	15,565	× 344
	Total			17.4	100		
Selected drainages, Coast Mountains, southwest British Columbia	Forest	32	88.9	246.1	29	11.2	× 1.0
	Clear-cut	32	9.5	26.4	18	24.5	× 2.2
	Road	32	1.5	4.2	11	282.5	× 25.2
	Total	–	–	276.7	58	–	–
H. J. Andrews Experimental Forest, western Cascade Range, Oregon	Forest	25	77.5	49.8	31	35.9	× 1.0
	Clear-cut	25	19.3	12.4	30	132.2	× 3.7
	Road	25	3.2	2.0	69	1772	× 49
	Total	–	–	64.2	130	–	–

Figure 4.13 Soil erosion on a bare field in Oxfordshire, central England.

Table 4.11 Annual rates of soil loss (tonnes per hectare) under different land-use types in eastern England. Source: From Morgan (1977)

Plot type	Slope location	Splash	Overland flow	Rill	Total
Bare soil	Top	0.33	6.67	0.10	7.10
	Middle	0.82	16.48	0.39	17.69
	Lower	0.62	14.34	0.06	15.02
Bare soil	Top	0.60	1.11	–	1.71
	Middle	0.43	7.78	–	8.21
	Lower	0.37	3.01	–	3.38
Grass	Top	0.09	0.09	–	0.18
	Middle	0.09	0.57	–	0.68
	Lower	0.12	0.05	–	0.17
Woodland	Top	–	–	–	0.00
	Middle	–	0.012	–	0.012
	Lower	–	0.008	–	0.008

2 Use of larger and heavier agricultural machinery, which has a tendency to increase soil compaction.
3 Removal of hedgerows and the associated increase in field size. Larger fields cause an increase in slope length with a concomitant increase in erosion risk.
4 Declining levels of organic matter resulting from intensive cultivation and reliance on chemical fertilizers, which in turn lead to reduced aggregate stability.
5 Availability of more powerful machinery, which permits cultivation in the direction of maximum slope rather than along the contour. Rills often develop along tractor and implement wheel tracks and along drill lines.

6 Use of powered harrows in seedbed preparation and the rolling of fields after drilling.
7 Widespread introduction of autumn-sown cereals to replace spring-sown cereals. Because of their longer growing season, winter cereals produce greater yields and are therefore more profitable. The change means that seedbeds are exposed with little vegetation cover throughout the period of winter rainfall.

Water is not the only active process creating accelerated erosion in eastern England, though it is important (Evans and Northcliff, 1978). Ever since the 1920s dust storms have been recorded in the Fenlands, the Brecklands, East Yorkshire (Radley and Sims, 1967), and Lincolnshire (see, e.g., Arber, 1946), and they seem to be occurring with increasing frequency. The storms result from changing agricultural practices, including the substitution of artificial fertilizers for farmyard manure, a reduction in the process of 'claying', whereby clay was added to the peat to stabilize it, the removal of hedgerows to facilitate the use of bigger farm machinery, and, perhaps most importantly, the increased cultivation of sugar beet. This crop requires a fine tilth and tends to leave the soil relatively bare in early summer compared with other crops (Pollard and Miller, 1968).

Nevertheless, possibly the most famous case of soil erosion by deflation was the Dust Bowl of the 1930s in the USA (Figure 4.14). In part this was caused by a series of hot, dry years which depleted the vegetation cover and made the soils dry enough to be susceptible to wind erosion. The effects of this drought were gravely exacerbated by years of overgrazing and unsatisfactory farming techniques. However, perhaps the prime cause of the event was the rapid expansion of wheat cultivation in the Great Plains. The number of cultivated hectares doubled during the First World War as tractors (for the first time) rolled out on to the plains by the thousands. In Kansas alone wheat hectarage increased from under 2 million hectares in 1910 to almost 5 million in 1919. After the war wheat cultivation continued apace, helped by the development of the combine harvester and government assistance. The farmer, busy sowing wheat and reaping gold, could foresee no end to his land of milk and honey, but the years of favorable climate were not to last, and over large areas the tough sod which exasperated the earlier homesteaders had given way to friable soils of high erosion potential. Drought, acting on damaged soils, created the 'black blizzards' that have been so graphically described by Coffey (1978).

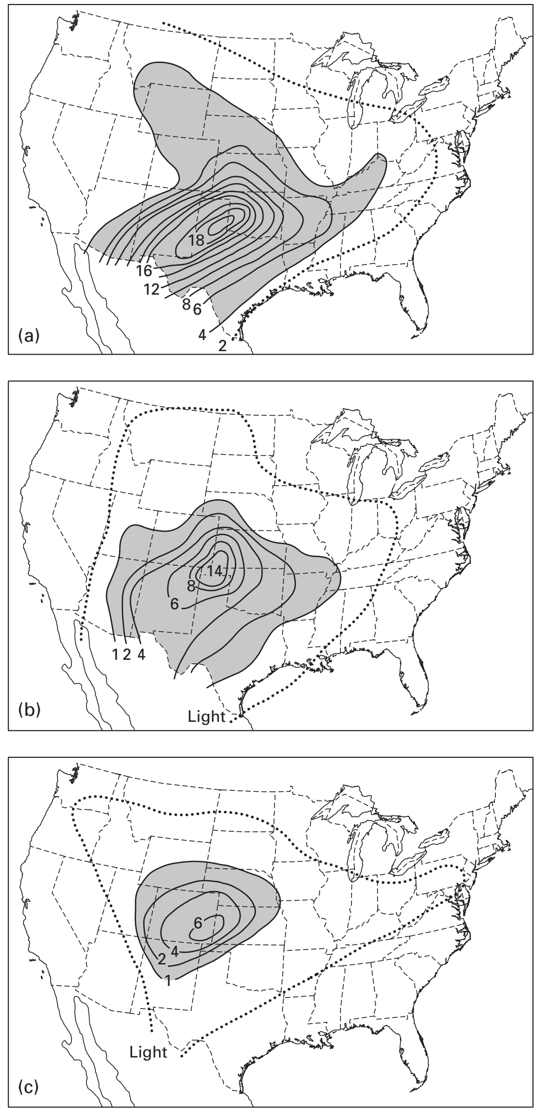

Figure 4.14 The concentration of dust storms (number of days per month) in the USA in 1939, illustrating the extreme localization over the High Plains of Texas, Colorado, Oklahoma, and Kansas: (a) March, (b) April, and (c) May (after Goudie, 1983).

Dust storms are still a serious problem in various parts of the USA; the Dust Bowl was not solely a feature of the 1930s (Figure 4.15). Thus, for example, in the San Joaquin Valley area of California in 1977 a dust storm caused extensive damage and erosion over an area of about 2000 km^2. More than 25 million tonnes

Figure 4.15 In the High Plains near Lubbock in Texas the effect of soil erosion and drifting was very evident in 1977. Note the vast fields and absence of windbreaks.

of soil were stripped from grazing land within a 24-hour period. While the combination of drought and a very high wind provided the predisposing natural conditions for the stripping to occur, overgrazing and the general lack of windbreaks in the agricultural land played a more significant role. In addition, broad areas of land had recently been stripped of vegetation, leveled or plowed up prior to planting. Other quantitatively less important factors included stripping of vegetation for urban expansion, extensive denudation of land in the vicinity of oilfields, and local denudation of land by vehicular recreation (Wilshire et al., 1981). One interesting observation made in the months after the dust storm was that in subsequent rainstorms runoff occurred at an accelerated rate from those areas that had been stripped by the wind, exacerbating problems of flooding and initiating numerous gullies. Elsewhere in California dust yield has been considerably increased by mining operations in dry lakebeds (Wilshire, 1980) and by disturbance of playas (Gill, 1996).

A comparable acceleration of dust storm activity occurred in the former Soviet Union. After the 'Virgin Lands' program of agricultural expansion in the 1950s, dust storm frequencies in the southern Omsk region increased on average by a factor of 2.5 and locally by factors of 5 to 6. Data on trends elsewhere are evaluated by Goudie (1983: 520) and Goudie and Middleton (1992). A good review of wind erosion of agricultural land is provided by Warren (2002).

Soil erosion produced by fire

Many fires are started by humans, either deliberately or non-deliberately, and because fires remove vegetation and expose the ground they tend to increase rates of soil erosion (see Table 4.12).

Table 4.12 Recent examples of fire-induced soil erosion

Source	*Location*
Wilson (1999)	Tasmania
Cerda (1998)	Mediterranean region
Moody and Martin (2001)	Colorado Front Range, USA
Dragovich and Morris (2002)	Eastern Australia
Pierson et al. (2002)	Idaho, USA

Table 4.13 Soil erosion associated with *Calluna* (heather) burning on the North Yorkshire Moors. Source: Data in Imeson (1971)

Condition of Calluna *or ground surface*	*Mean rate of litter accumulation (+) or erosion (–) (mm year^{-1})*	*Number of observations*
Calluna 30–40 cm high. Complete canopy	+3.81	60
Calluna 20–30 cm high. Complete canopy	+0.25	20
Calluna 15–20 cm high. 40–100% cover	−0.74	20
Calluna 5–15 cm high. 10–100% cover	−6.4	20
Bare ground. Surface of burnt *Calluna*	−9.5	19
Bare ground. Surface of peaty or mineral subsoil	−45.3	25

The burning of forests, for example, can, especially in the first years after the fire event, lead to high rates of soil loss. Burnt forests often have rates a whole order of magnitude higher than those of protected areas. Comparably large changes in soil erosion rates have been observed to result from the burning of heather in the Yorkshire moors in northern England (see Table 4.13), and the effects of burning may be felt for the six years or more that may be required to regenerate the heather (*Calluna*). In the Australian Alps fire in experimental catchments has been found to lead to a greatly increased flow in the streams, together with a marked surge in the delivery of suspended load. Combining the two effects of increased flow rate and sediment yield, it was found that, after fire, the total sediment load was increased 1000 times (Pereira, 1973). Likewise, watershed experiments in the chaparral scrub of Arizona, involving denudation by a destructive fire, indicated that whereas erosion losses before the fire were only 43 tonnes per square kilometer per year, after the fire they were between 50,000 and 150,000 tonnes per square kilometer per year. The causes of the marked erosion associated with chaparral burning are particularly interesting. There is normally a distinctive 'non-wettable' layer in the soils supporting chaparral. This layer, composed of soil particles coated by hydrophobic substances leached from the shrubs or their litter, is normally associated with the upper part of the soil profile (Mooney and Parsons, 1973), and builds up through time in the unburned chaparral. The high temperatures that accompany chaparral fires cause these hydrophobic substances to be distilled so that they condense on lower soil layers. This process results in a shallow layer of wettable soil overlying a non-wettable layer. Such a condition, especially on steep slopes, can result in severe surface erosion (DeBano, 2000; Shakesby et al., 2000; Letey, 2001).

In chaparral terrain it is possible to envisage a fire-induced sediment cycle (Graf, 1988: 243). It starts with a fire that destroys the scrub and the root net, and changes surface soil properties in the way already discussed. After the fire, a precipitation event of low magnitude (with a return interval of around one or two years) is sufficient to induce extensive sheet and rill erosion, which removes enough soil to retard vegetation recovery. Eventually, a larger precipitation event occurs (with a return interval of around five to ten years) and, because of limited vegetation cover, produces severe debris slides. Slowly the vegetation cover re-establishes itself, and erosion rates diminish. However, in due course enough vegetation grows to create a fire hazard, and the whole process starts again.

Soil erosion associated with construction and urbanization

There are now a number of studies which illustrate clearly that urbanization can create significant changes in erosion rates.

The highest rates of erosion are produced in the construction phase, when there is a large amount of exposed ground and much disturbance produced by vehicle movements and excavations. Wolman and Schick (1967) and Wolman (1967) have shown that the equivalent of many decades of natural or even agricultural erosion may take place during a single year in areas cleared for construction. In Maryland they found that sediment yields during construction reached 55,000 tonnes per square kilometer per year, while in the same area rates under forest were around 80–200 tonnes per square kilometer per year and those under farm 400 tonnes per square kilometer per year. New road cuttings in Georgia were found to have sediment yields up to 20,000–50,000 tonnes per square kilometer per year. Likewise, in Devon, England, Walling and Gregory (1970) found that suspended sediment concentrations in streams draining construction areas were two to ten times (occasionally up to 100 times) higher than those in undisturbed areas. In Virginia, USA, Vice et al. (1969) noted equally high rates of erosion during construction and reported that they were ten times those from agricultural land, 200 times those from grassland, and 2000 times those from forest in the same area.

However, construction does not go on forever, and once the disturbance ceases, roads are surfaced, and gardens and lawns are cultivated. The rates of erosion fall dramatically and may be of the same order as those under natural or pre-agricultural conditions (Table 4.14). Moreover, even during the construction phase several techniques can be used to reduce sediment removal, including the excavation of settling ponds, the seeding and mulching of bare surfaces, and the erection of rock dams and straw bales (Reed, 1980).

Attempts at soil conservation

Because of the adverse effects of accelerated erosion a whole array of techniques has now been widely adopted to conserve soil resources (Hudson, 1987). Some of the techniques such as hillslope terracing may be of some antiquity, and traditional techniques have both a wide range of types and also many virtues (see Critchley et al., 1994; Reij et al., 1996). The following are some of the main ways in which soil cover may be conserved.

1 Revegetation:
   - deliberate planting;
   - suppression of fire, grazing, etc., to allow regeneration.
2 Measures to stop stream bank erosion.

Table 4.14 Rates of erosion associated with construction and urbanization

Location	*Land use*	*Source*	*Rate (t km^{-2} yr^{-1})*
Maryland, USA	Forest	Wolman (1967)	39
	Agriculture		116–309
	Construction		38,610
	Urban		19–39
Virginia, USA	Forest	Vice et al. (1969)	9
	Grassland		94
	Cultivation		1876
	Construction		18,764
Detroit, USA	General non-urban	Thompson (1970)	642
	Construction		17,000
	Urban		741
Maryland, USA	Rural	Fox (1976)	22
	Construction		37
	Urban		337
Maryland, USA	Forest and grassland	Yorke and Herb (1978)	7–45
	Cultivated land		150–960
	Construction		1600–22,400
	Urban		830
Wisconsin, USA	Agricultural	Daniel et al. (1979)	< 1
	Construction		19.2
Tama New Town, Japan	Construction	Kadomura (1983)	*c.* 40,000
Okinawa, Japan	Construction	Kadomura (1983)	25,000–125,000

3 Measures to stop gully enlargement:
- planting of trailing plants, etc.;
- weirs, dams, gabions, etc.

4 Crop management:
- maintaining cover at critical times of year;
- rotation;
- cover crops.

5 Slope runoff control:
- terracing;
- deep tillage and application of humus;
- transverse hillside ditches to interrupt runoff;
- contour plowing;
- preservation of vegetation strips (to limit field width).

6 Prevention of erosion from point sources such as roads and feedlots:
- intelligent geomorphic location;
- channeling of drainage water to nonsusceptible areas;
- covering of banks, cuttings, etc., with vegetation.

7 Suppression of wind erosion:
- soil moisture preservation;
- increase in surface roughness through plowing up clods or by planting windbreaks.

An alternative way of classifying soil conservation is provided by Morgan (1995). He identifies three main types of measure: agronomic, soil management, and mechanical. The effect of these in relation to the main detachment and transport phases of erosion are shown in Table 4.15.

There are some parts of the world where terraces (a mechanical measure) are one of the most prominent components of the landscape. This applies to many wine-growing areas, to some arid zone regions (such as southwest USA, Yemen, and Peru), and to a wide selection of localities in the more humid tropics (Luzon, Java, Sumatra, Assam, Ceylon, Uganda, Cameroons, the Andes, etc.). However, much traditional terracing only had soil erosion control as a secondary motive.

Table 4.15 Effect† of various soil conservation practices on the detachment (D) and transport (T) phases of erosion. Source: Morgan (1995, table 7.1)

Practice	*Rainsplash*		*Runoff*		*Wind*	
	D	*T*	*D*	*T*	*D*	*T*
Agronomic measures:						
(a) covering soil surface	*	*	*	*	*	*
(b) increasing surface roughness	–	–	*	*	*	*
(c) increasing surface depression storage	+	+	*	*	–	–
(d) increasing infiltration	–	–	+	*	–	–
Soil management:						
(a) fertilizers, manures	+	+	+	*	+	*
(b) subsoiling, drainage	–	–	+	*	–	–
Mechanical measures:						
(a) contouring, ridging	–	+	+	*	+	*
(b) terraces	–	+	+	*	–	–
(c) shelterbelts	–	–	–	–	*	*
(d) waterways	–	–	–	–	*	–

†– no control; + moderate control; *strong control.

More often these were constructed to provide level planting surfaces, to provide deeper soil, and to manage the flow of water (Doolittle, 2000). In areas subject to wind erosion other strategies may be necessary. Since soil only blows when it is dry, anything which conserves soil moisture is beneficial. Another approach to wind erosion is to slow down the wind by physical barriers, either in the form of an increased roughness of the soil surface brought about by careful plowing, or by planted vegetative barriers, such as windbreaks and shelterbelts.

Some attempts at soil conservation have been particularly successful. For example, in Wisconsin, a study by Trimble and Lund (1982) showed that in the Coon Creek Basin erosion rates declined fourfold between the 1930s and the 1970s. One of the main reasons for this was the progressive adoption of contour-strip plowing (Figure 4.16).

Attempts at soil conservation have not always been without their drawbacks. For example, the establishment of ground cover in dry areas to limit erosion may so reduce soil moisture because of accelerated

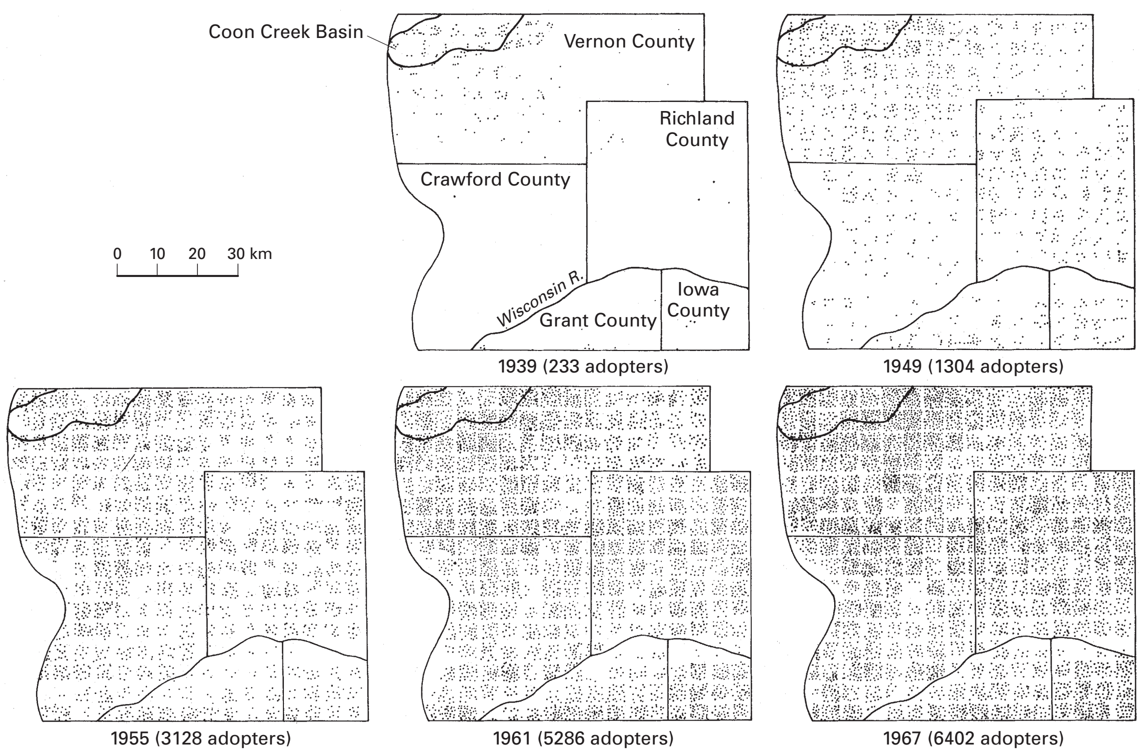

Figure 4.16 The spread of contour-strip soil conservation methods in Wisconsin, USA, between 1939 and 1967. One dot represents one adopter (after H. E. Johansen, in Trimble and Lund, 1982, figure 22).

evapotranspiration that the growth of the main crop is adversely affected. On a wider scale major afforestation schemes can cause substantial runoff depletion in river catchments. Likewise, the provision of mulching is sometimes detrimental: in cool climates, reduced soil temperature shortens the growing season, while in wet areas, higher soil moisture may induce gleying and anaerobic conditions (Morgan, 1979: 60). Some terrace schemes have also had their shortcomings. They have been known to hold back so much water on hillsides that the soils have become saturated and landsliding has been induced. Similarly strip cropping, because it involves the farming of small areas, is incompatible with highly mechanized agricultural systems, and insect infestation and weed control are additional problems which it has posed.

Soil conservation measures may not always be appropriate, for, paradoxically, soil erosion can be a useful phenomenon. Sanchez and Buol (1975), for instance, have pointed out that in recent volcanic areas soil erosion has enabled removal of the more weathered base-depleted material from the soil surface, exposing the more fertile, less weathered, base-rich material beneath. Likewise, in the Nochixtland area of southern Mexico, soil erosion has been utilized by local farmers to *produce* agricultural land. Severe gullies have cut into steep valley-side slopes, and since the Spanish Conquest an average depth of 5 m has been stripped from the entire surface area. The local Mixtec farmers, far from seeing this high rate of erosion as a hazard to be feared, have directed the flow of the eroded material to feed their fields with fertile soil and to extend their land. Over the past 1000 years (see Whyte, 1977), the Mixtec cultivators have managed to use gully erosion to double the width of the main valley floors with flights of terraces. Judicious use of the phenomenon of gully erosion has enabled them to convert poor hilltop fields into the rich alluvial farmland below.

Nonetheless it is undoubtedly true that manipulation of the soil is one of the most significant ways in which humans change the environment, and one in which they have had some of the most detrimental effects. Soil deterioration has led to many cases of what W. C. Lowdermilk once termed 'regional suicide', and the overall situation is at least as bleak as it was in the post-Dust Bowl years when Jacks commented: 'The organization of civilized societies is founded upon the measures taken to wrest control of the soil from wild Nature, and not until complete control has passed into human hands can a stable super-structure of what we call civilization be erected on the land' (Jacks and Whyte, 1939: 17).

Points for review

What are the causes of accelerated salinization?
Do humans contribute to lateritization and podzolization?
How does the removal of forest cover affect soil erosion?
What effects do fires have on soils?
How might you seek to reduce rates of soil erosion by (a) water, and (b) wind?
How serious a risk is soil erosion to agricultural sustainability?

Guide to reading

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Fullen, M. A. and Catt, J. A., 2004, *Soil management: problems and solutions*. A review of some of the problems involved in the management of soils in the face of human impacts.

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McTainsh, G. and Boughton, W. C., 1993, *Land degradation processes in Australia*. Melbourne: Longman Cheshire. An Australian perspective on soil modification.

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