2 THE HUMAN IMPACT ON VEGETATION

Introduction

In any consideration of the human impact on the environment it is probably appropriate to start with vegetation, for humankind has possibly had a greater influence on plant life than on any of the other components of the environment. Through the many changes humans have brought about in land use and land cover they have modified soils (Meyer and Turner 1994) (see Chapter 4), influenced climates (see Chapter 7), affected geomorphic processes (see Chapter 6), and changed the quality (see Chapter 5) and quantity of some natural waters. Indeed, the nature of whole landscapes has been transformed by human-induced vegetation change (Figure 2.1). Hannah et al. (1994) attempt to provide a map and inventory of human

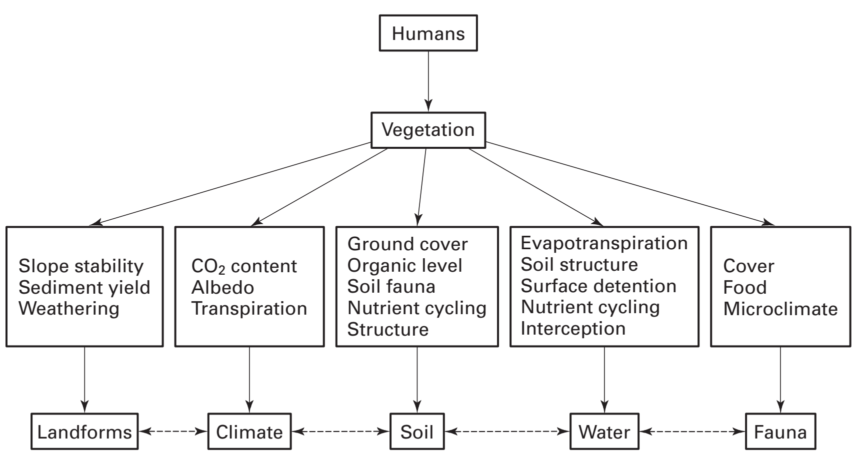

Figure 2.1 Some ramifications of human-induced vegetation change.

disturbance of world ecosystems, but many of their criteria for recognizing undisturbed areas are naïve. Large tracts of central Australia are classed as undisturbed, when plainly they are not (Fitzpatrick, 1994). Following Hamel and Dansereau (1949, cited by Frenkel, 1970) we can recognize five principal degrees of interference – each one increasingly remote from pristine conditions. These are:

1 *Natural habitats*: those that develop in the absence of human activities.
2 *Degraded habitats*: those produced by sporadic, yet incomplete, disturbances; for example, the cutting of a forest, burning and the non-intensive grazing of natural grassland.
3 *Ruderal habitats*: where disturbance is sustained but where there is no intentional substitution of vegetation. Roadsides are an example of a ruderal habitat.
4 *Cultivated habitats*: when constant disturbance is accompanied by the intentional introduction of plants.
5 *Artificial habitats*: which are developed when humans modify the ambient climate and soil, as in greenhouse cultivation.

An alternative model for classifying the extent of human influence on vegetation is provided by Westoff (1983), who adopts a four-part scheme:

1 *Natural*: a landscape or an ecosystem not influenced by human activity.
2 *Subnatural*: a landscape or ecosystem partly influenced by humans, but still belonging to the same (structural) formation type as the natural system from which it derives (e.g., a wood remaining a wood).
3 *Semi-natural*: a landscape or ecosystem in which flora and fauna are largely spontaneous, but the vegetation structure is altered so that it belongs to another formation type (e.g., a pasture, moorland or heath deriving from a wood).
4 *Cultural*: a landscape or ecosystem in which flora and fauna have been essentially affected by human agency in such a way that the dominant species may have been replaced by other species (e.g., arable land).

In this chapter we shall be concerned mainly with degraded and ruderal habitats, or subnatural and semi-natural habitats, but first we need to consider some of the processes that human societies employ: notably fire, grazing and the physical removal of forest.

The use of fire

Humans are known to have used fire since Paleolithic times (see Chapter 1). As Sauer, one of the great proponents of the role of fire in environmental change, put it (1969: 10–11):

> Through all ages the use of fire has perhaps been the most important skill to which man has applied his mind. Fire gave to man, a diurnal creature, security by night from other predators . . . The fireside was the beginning of social living, the place of communications and reflection.

People have utilized fire for a great variety of reasons (Bartlett, 1956; Stewart, 1956; Wertine, 1973): to clear forest for agriculture, to improve grazing land for domestic animals or to attract game; deprive game of cover; drive game from cover in hunting; kill or drive away predatory animals, ticks, mosquitoes, and other pests; repel the attacks of enemies, or burn them out of their refuges; cooking; expedite travel; burn the dead and raise ornamental scars on the living; provide light; transmit messages via smoke signaling; break up stone for tool-making: protect settlements or encampments from great fires by controlled burning; satisfy the sheer love of fires as spectacles; make pottery; smelt ores; harden spears; provide warmth; make charcoal; and assist in the collection of insects such as crickets for eating. Given this remarkable utility, it would be surprising if it had not been turned to account. Indeed it is still much used, especially by pastoralists such as the cattle-keepers of Africa, and by shifting agriculturists. For example, the Malaysian and Indonesian *ladang* and the *milpa* system of the Maya in Latin America involved the preparation of land for planting by felling or deadening forest, letting the debris dry in the hot season, and burning it before the commencement of the rainy season. With the first rains, holes were dibbled in the soft ash-covered earth with a planting stick. This system was suited to areas of low population density with sufficiently extensive forest to enable long intervals of 'forest fallow' between burnings. Land that was burned too frequently became overgrown with perennial grasses, which tended to make it difficult to farm with primitive tools. Land cultivated for too long rapidly suffered deterioration in fertility, while land recently burned was temporarily rich in nutrients (see Figure 13.1).

The use of fire, however, has not been restricted to primitive peoples in the tropics. Remains of charcoal are found in Holocene soil profiles in Britain (Moore, 2000); large parts of North America appear to have suffered fires at regular intervals prior to European settlement (Parshall and Foster, 2002); and in the case of South America the 'great number of fires' observed by Magellan during the historical passage of the Strait that bears his name resulted in the toponym, 'Tierra del Fuego'. Indeed, says Sternberg (1968: 718): 'for thousands of years, man has been putting the New World to the torch, and making it a "land of fire".' Given that the native American population may have been greater than once thought, their impact, even in Amazonia, may have been appreciable (Deneven, 1992). North American Indians were also not passive occupants of the land. They managed forests extensively creating 'a dynamic mosaic containing patches of young and old trees, interspersed with patches of grass and shrubs' (Bonnicksen, 1999: 442). As a result, some North American forests, prior to European settlement, were open and park-like, not dark and dense.

Fire (Figure 2.2) was also central to the way of life of the Australian aboriginals, including those of Tasmania (Hope, 1999), and the carrying of fire sticks was a common phenomenon. As Blainey (1975: 76) has put it, 'Perhaps never in the history of mankind was there a people who could answer with such unanimity the question: "have you got a light, mate?". There can have been few if any races who for so long were able to practise the delights of incendiarism.' That is not to say that aboriginal burning necessarily caused wholesale modification of Australian vegetation in pre-European times. That is a hotly debated topic (Kohen, 1995).

Figure 2.2 The pindan bush of northwest Australia, composed of *Eucalyptus*, *Acacia*, and grasses, is frequently burned. The pattern of the burning shows up clearly on Landsat imagery.

In neighboring New Zealand, Polynesians carried out extensive firing of vegetation in pre-European settlement times, and hunters used fire to facilitate travel and to frighten and trap a major food source – the flightless moa (Cochrane, 1977). The changes in vegetation that resulted were substantial (Figure 2.3). The forest cover was reduced from about 79% to 53%, and fires were especially effective in the drier forests of central and eastern South Island in the rain shadow of the Southern Alps. The fires continued over a period of about 1000 years up to the period of European settlement (Mark and McSweeney, 1990). Pollen analyses (Figure 2.4) show dramatically the reduction in tall trees and the spread of open shrubland following the introduction of the widespread use of fire by the Maoris (McGlone and Wilmshurst, 1999).

Fires: natural and anthropogenic

Although people have used fire for all the reasons that have been mentioned, before one can assess the role of this facet of the human impact on the environment one must ascertain how important fires started by human action are in comparison with those caused

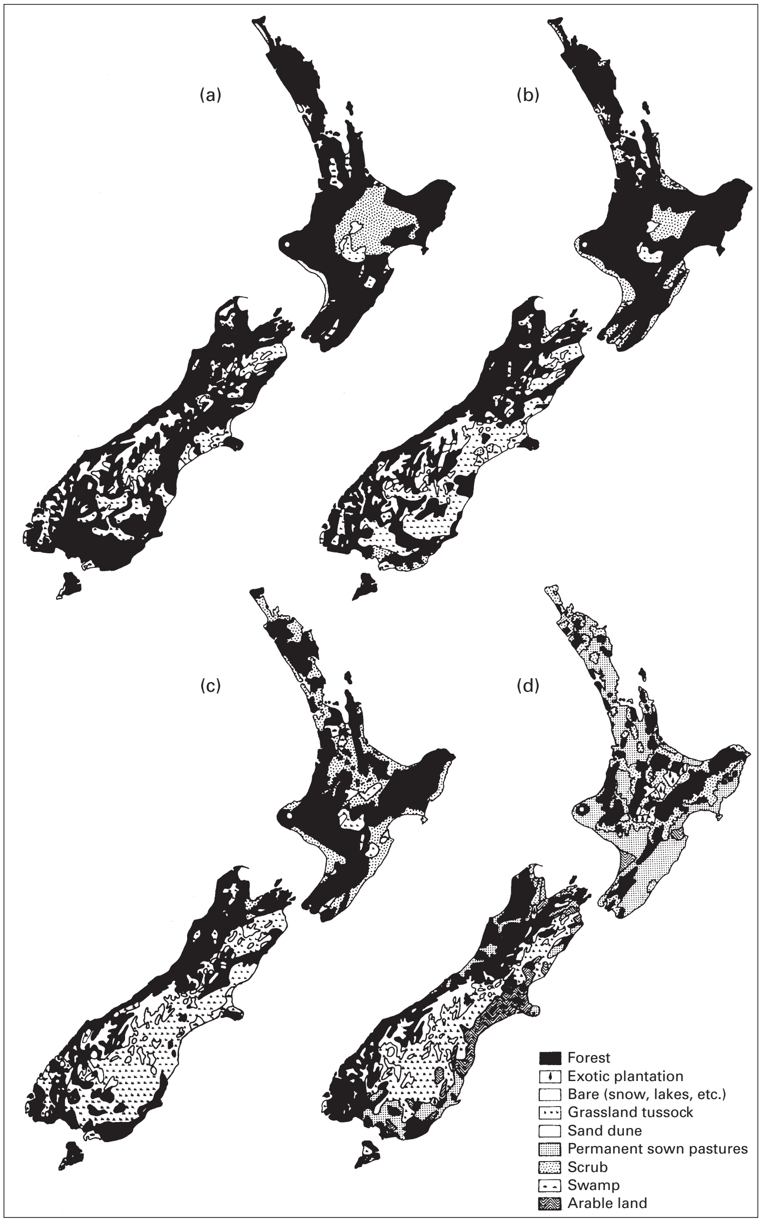

Figure 2.3 The changing state of the vegetation cover in New Zealand (from Cochrane, 1977): (a) early Polynesian vegetation, *c.* AD 700; (b) Pre-Classical Maori vegetation, *c.* AD 1200; (c) Pre-European vegetation, *c.* AD 1800; (d) present-day vegetation.

naturally, especially by lightning, which on average strikes the land surface of the globe 100,000 times each day (Yi-Fu Tuan, 1971). Some natural fires may result from spontaneous combustion (Vogl, 1974), for in certain ecosystems heavy vegetal accumulations may become compacted, rotted and fermented, thus generating heat. Other natural fires can result from sparks produced by falling boulders and by landslides (Booysen and Tainton, 1984). In the forest lands of the western USA about half the fires are caused by lightning. Lightning starts over half the fires in the pine savanna of Belize, Central America, and about 8% of the fires in the bush of Australia, while in the south of France nearly all the fires are caused by people.

One method of gauging the long-term frequency of fires (Clarke et al., 1997) is to look at tree rings and lake sediments, for these are affected by them. Studies in the USA indicate that over wide areas fires occurred

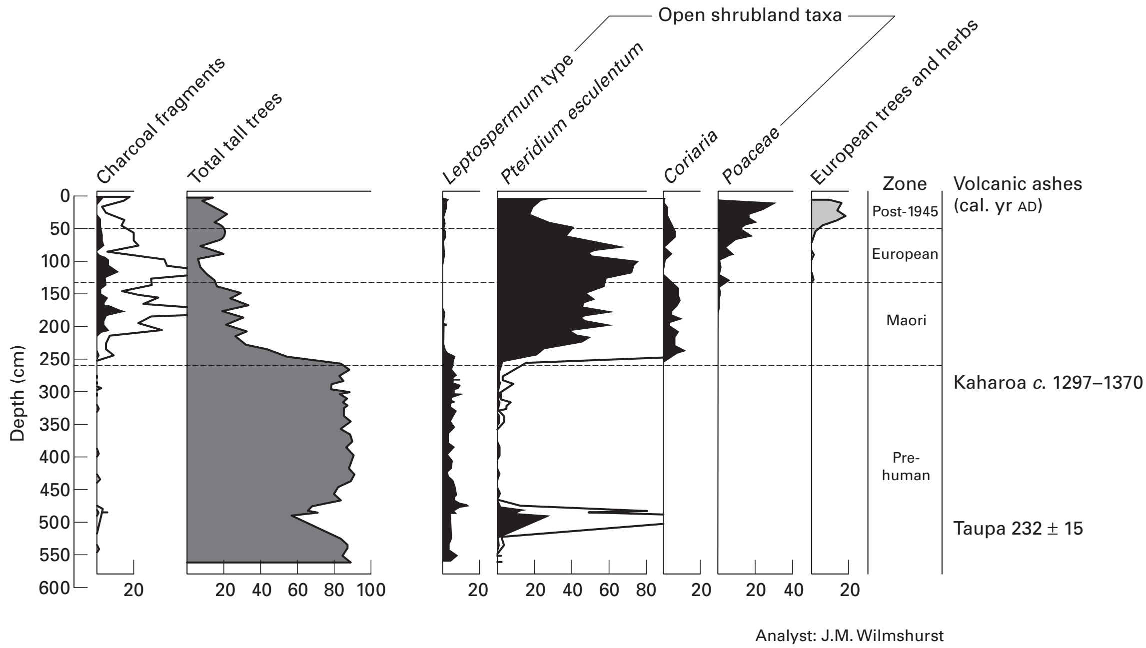

Figure 2.4 Summary percentage pollen diagram, Lake Rotonuiaha, eastern North Island (from McGlone and Wilmshurst, 1999, figure 1), showing the changes in vegetation brought about in Maori and European times.

with sufficient frequency to have effects on annual tree rings and lake cores every 7 to 80 years in pre-European times. The frequency of fires in different environments tends to show some variation. Rotation periods may be in excess of a century for tundra, 60 years for boreal pine forest, 100 years for spruce-dominated ecosystems, 5 to 15 years for savanna, 10 to 15 years for chaparral, and less than 5 years for semi-arid grasslands (Wein and Maclean, 1983).

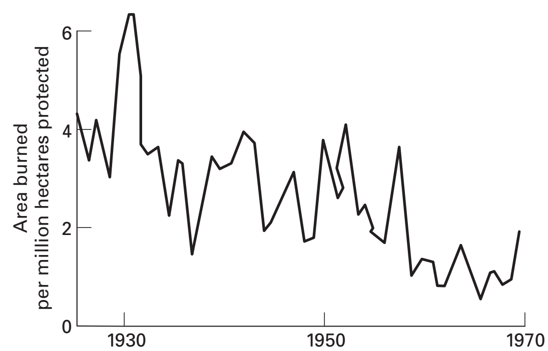

Figure 2.5 The reduction in area burned per million hectares protected for the USA between 1926 and 1969 as a result of fire-suppression policies (after Brown and Davis, 1973, figure 2.1).

The temperatures attained in fires

The effects which fires have on the environment depend very much on their size, duration, and intensity. Some fires are relatively quick and cool, and destroy only ground vegetation. Other fires, crown fires, affect whole forests up to crown level and generate very high temperatures In general forest fires are hotter than grassland fires. Perhaps more significantly in terms of forest management, fires that occur with great frequency do not attain too high a temperature because there is inadequate inflammable material to feed them. However, when humans deliberately suppress fire, as has frequently been normal policy in forest areas (see Figure 2.5), large quantities of inflammable materials accumulate, so that when a fire does break out it is of the hot, crown type. Such fires can be disastrous ecologically and there is now much debate about the wisdom of fire suppression given that, in many forests, fires under so-called ‘natural’ conditions appear, as we have seen, to have been a relatively frequent and regular phenomenon.

Some consequences of fire suppression

Given that fire has long been a feature of many ecosystems, and irrespective of whether the fires were or were not caused by people, it is clear that any deliberate policy of fire suppression will have important consequences for vegetation.

Fire suppression, as has already been suggested, can magnify the adverse effects of fire. The position has been well stated by Sauer (1969: 14):

> The great fires we have come to fear are effects of our civilization. These are the crown fires of great depths and heat, notorious aftermaths of the pyres of slash left by lumbering. We also increase fire hazard by the very giving of fire protection, which permits the indefinite accumulation of inflammable litter. Under the natural and primitive order, such holocausts, that leave a barren waste, even to the destruction of the organic soil, were not common.

Recent studies have indicated that rigid fire-protection policies have often had undesirable results (Bonnicksen et al., 1999) and as a consequence many foresters stress the need for 'environmental restoration burning' (Vankat, 1977). One of the best ways to prevent the largest forest fires is to allow the small and medium fires to burn (Malamud et al., 1998). So, for example, in the coniferous forests of the middle upper elevations in the Sierra Nevada mountains of California fire protection since 1890 has made the stands denser, shadier and less park-like, and sequoia seedlings have decreased in number. Likewise, at lower elevations the Mediterranean semi-arid shrubland, called chaparral, has had its character changed. The vegetation has increased in density, the amount of combustible fuels has risen, fire-intolerant species have encroached, and vegetation diversity has decreased, resulting in a monotony of old-age stands, instead of a mosaic of different successional stages. Unfavorable consequences of fire suppression have also been noted in Alaska (Oberle, 1969). It has been found that when fire is excluded from many lowland sites an insulating carpet of moss tends to accumulate and raise the permafrost level. **Permafrost** close to the surface encourages the growth of black spruce, a low-growing species with little timber or food value. In the Kruger National Park, in South Africa, fires have occurred less frequently after the establishment of the game reserve, when it became uninhabited by natives and hunters. As a result, bush encroachment has taken place in areas that were formerly grassland and the carrying capacity for grazing animals has declined. Controlled burning has been re-instituted as a necessary game-management operation.

However, following the severe fires that ravaged America's Yellowstone National Park in 1988, there has been considerable debate as to whether the inferno, the worst since the park was established in the 1870s, was the result of a policy of fire suppression. Without such a policy the forest would burn at intervals of 10–20 years because of lightning strikes. Could it be that the suppression of fires over long periods of, say, 100 years or more, allegedly to protect and preserve the forest, led to the build-up of abnormal amounts of combustible fuel in the form of trees and shrubs in the understory? Should a program of prescribed burning be carried out to reduce the amount of fuel available?

Fire suppression policies at Yellowstone did indeed lead to a critical build-up in flammable material. However, other factors must also be examined in explaining the severity of the fire. One of these was the fact that the last comparable fire had been in the 1700s, so that the Yellowstone forests had had nearly 300 years in which to become increasingly flammable. In other words, because of the way vegetation develops through time (a process called **succession**) very large fires may occur every 200–300 years as part of the natural order of things (Figure 2.6). Another crucial factor was that the weather conditions in the summer of 1988 were abnormally dry, bringing a great danger of fire. As Romme and Despain (1989: 28) remarked:

> It seems that unusually dry, hot and windy weather conditions in July and August of 1988 coincided with multiple ignitions in a forest that was at its most flammable stage of succession. Yet it is unlikely that past suppression efforts greatly exacerbated the Yellowstone fire. If fires occur naturally at intervals ranging from 200 to 400 years, then 30 or 40 years of effective suppression is simply not enough for excessive quantities of fuel to build up. Major attempts at suppression in the Yellowstone forests may have merely delayed the inevitable.

Certainly, studies of the long-term history of fire in the western USA have shown the great importance of climatic changes at the decennial and centennial scales in controlling fire frequency during the Holocene (Meyer and Pierce, 2003; Whitlock et al., 2003).

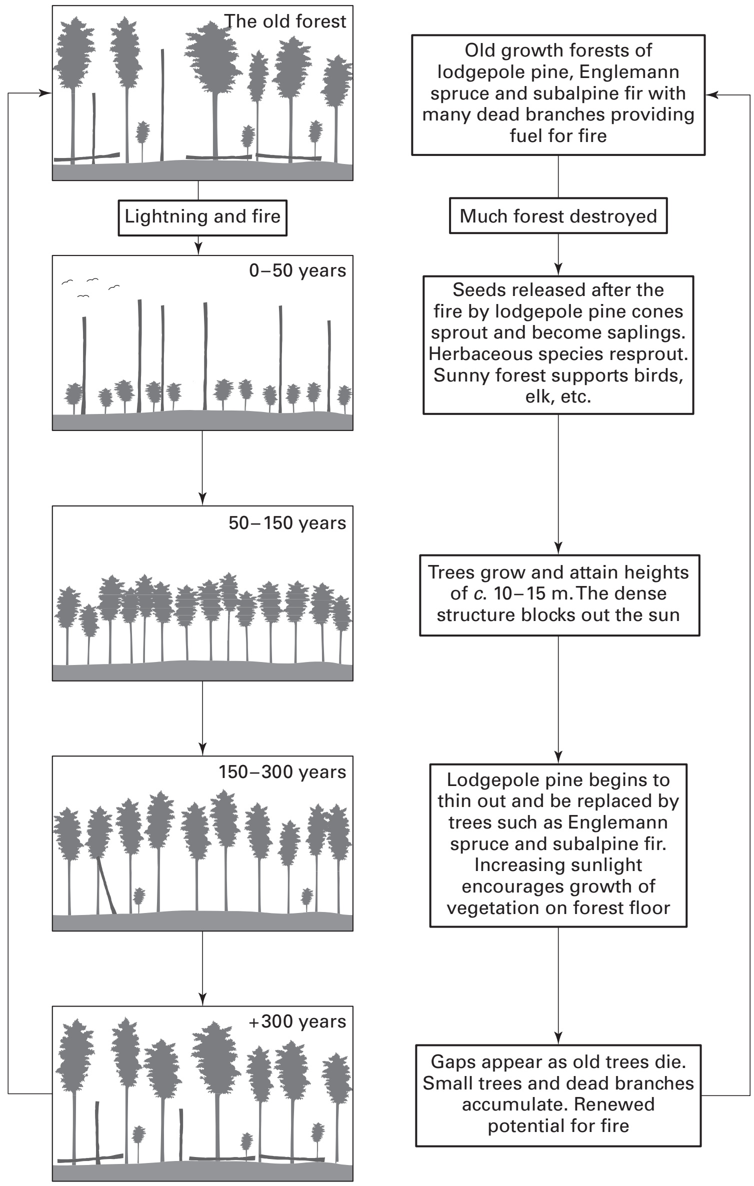

Figure 2.6 Ecological succession in response to fire in Yellowstone National Park (after Romme and Despain, 1989: 24–5, greatly modified).

Some effects of fire on vegetation

There is evidence that fire has played an important role in the formation of various major types of vegetation. This applies, for instance, to some tropical savannas, and mid-latitude grasslands and shrublands. Before examining these, however, it is worth looking at some of the general consequences of burning (Crutzen and Goldammer, 1993).

Fire may assist in seed germination. For example, the abundant germination of dormant seeds on recently burned **chaparral** sites has been reported by many investigators, and it seems that some seeds of chaparral species require scarification by fire. The better germination of those not requiring scarification may be related to the removal by fire of competition, litter and some substances in the soils which are toxic to plants (Hanes, 1971). Fire alters seedbeds. If litter and humus removals are substantial, large areas of rich ash, bare soil, or thin humus may be created. Some trees, such as Douglas fir and the giant sequoia, benefit from such seedbeds (Heinselman and Wright, 1973). Fire sometimes triggers the release of seeds (as with the Jack pine, *Pinus bankdiana*) and seems to stimulate the vegetative reproduction of many woody and herbaceous species. Fire can control forest insects, parasites and fungi – a process termed 'sanitization'. It also seems to stimulate the flowering and fruiting of many shrubs and herbs, and to modify the physicochemical environment of the plants. Mineral elements are released both as ash and through increased decomposition rates of organic layers. Above all, areas subject to fire often show greater species diversity, which is a factor that tends to favor stability.

One can conclude by quoting at length from Pyne (1982: 3), who provides a detailed and scholarly analysis of the history of cultural fires in America:

> Hardly any plant community in the temperate zone has escaped fire's selective action, and, thanks to the radiation of *Homo sapiens* throughout the world, fire has been introduced to nearly every landscape on earth. Many biotas have consequently so adapted themselves to fire that, as with biotas frequented by floods and hurricanes, adaptation has become symbiosis. Such ecosystems do not merely tolerate fire, but often encourage it and even require it. In many environments fire is the most effective form of decomposition, the dominant selective force for determining the relative distribution of certain species, and the means for effective nutrient recycling and even the recycling of whole communities.

The role of grazing

Many of the world's grasslands have long been grazed by wild animals, such as the bison of North America or the large game of East Africa, but the introduction of pastoral economies also affects their nature and productivity (Figure 2.7) (Coupland, 1979).

Light grazing may increase the productivity of wild pastures (Warren and Maizels, 1976). Nibbling, for example, can encourage the vigor and growth of plants, and in some species, such as the valuable African grass, *Themeda triandra*, the removal of coarse, dead stems permits succulent sprouts to shoot. Likewise the seeds of some plant species are spread efficiently by being carried in cattle guts, and then placed in favorable seedbeds of dung or trampled into the soil surface. Moreover, the passage of herbage through the gut and out as feces modifies the nitrogen cycle, so that grazed pastures tend to be richer in nitrogen than ungrazed ones. Also, like fire, grazing can increase species diversity by opening out the community and creating more niches.

Figure 2.7 Grazing by domestic animals has many environmental consequences and assists in the maintenance of grasslands. Excessive grazing can compact the soil and contribute to both wind and water erosion.

On the other hand, heavy grazing may be detrimental. Excessive trampling when conditions are dry will reduce the size of soil aggregates and break up plant litter to a point where they are subject to removal by eolian **deflation** processes. Trampling, by puddling the soil surface, can accelerate soil deterioration and erosion as infiltration capacity is reduced. Heavy grazing can kill plants or lead to a marked reduction in their level of photosynthesis. In addition, when relieved of competition from palatable plants or plants liable to trampling damage, resistant and usually unpalatable species expand their cover. Thus in the western USA poisonous burroweed (*Haploplappius* spp.) has become dangerously common, and many woody species have intruded. These include the mesquite (*Prosopis juliflora*), the big sagebrush (*Artemisia tridentata*) (Vale, 1974), the one-seed juniper (*Juniperus monosperma*) (Harris, 1966), and the Pinyon pine (Blackburn and Tueller, 1970).

Grover and Musick (1990) see shrubland encroachment by creosote bush (*Larrea tridentata*) and mesquite as part and parcel of desertification and indicate that in southern New Mexico the area dominated by them has increased several fold over the past century. This has been as a result of a corresponding decrease in the area's coverage of productive grasslands. They attribute both tendencies to excessive livestock overgrazing at the end of the nineteenth century, but point out that this was compounded by a phase of rainfall regimes that were unfavorable for perennial grass growth.

It is evident then that the semi-arid grasslands of southwestern North America have changed dramatically over the past 150 years as a result of the encroachment of native woody species such as mesquite. Van Auken (2000: 207) has concluded: 'The major cause of the encroachment of these woody species seems to be the reduction of grass **biomass** (fire fuel) by chronic high levels of domestic herbivory coupled to a reduction of grassland fires, which would have killed or suppressed the woody plants to the advantage of the grasses.'

There are many reasons why excessive grazing can lead to shrub dominance (Archer et al., 1999):

1 Livestock may effectively disperse woody plant seeds, particularly those of some leguminous shrubs and arborescents.
2 Utilization of grasses increases chances for germination and early establishment of woody seedlings.
3 Concomitant reductions in transpirational leaf area, root biomass, and root activity associated with grazing of grasses can:
   (a) increase superficial soil moisture to enhance woody seedling establishment and growth of shallow-rooted woody species;
   (b) increase the amount of water percolating to deeper depths and benefit established woody species with deep root systems;
   (c) increase nutrient availability to woody plants;
   (d) release suppressed populations of established tree or shrub 'seedling reserves'.
4 Grazing increases mortality rates and decreases plant basal area, seed production, and seedling establishment of palatable grasses.
5 Grazing may also increase susceptibility of grasses to other stresses such as drought. These factors would combine to increase the rate of gap formation and available area for woody plant seedling establishment, especially in post-drought periods.
6 Herbaceous species may be replaced by assemblages that compete less effectively with woody plants.
7 Reduction of the biomass and continuity of fine fuel may reduce fire frequency and intensity.
8 Invading woody species are often unpalatable relative to grasses and forbs and are thus not browsed with sufficient regularity or severity to limit establishment or growth.
9 Lower soil fertility and alterations in physicochemical properties occur with loss of ground cover and subsequent erosion. This favors N_2-fixing woody plants (e.g., *Prosopis*, *Acacia*) and evergreen woody plant growth forms that are tolerant of low nutrient conditions.

In Australia the widespread adoption of sheep grazing led to significant changes in the nature of grasslands over extensive areas. In particular, the introduction of sheep led to the removal of kangaroo grass (*Themeda australis*) – a predominantly summer-growing species – and its replacement by essentially winter-growing species such as *Danthonia* and *Stipa*. Also in Australia, not least in the areas of tropical savanna in the north, large herds of introduced feral animals (e.g., *Bos taurus, Equus caballus, Camelus dromedarius, Bos banteng*, and *Cervus unicolor*) have resulted in overgrazing and alteration of native habitats. As they appear to lack significant control by predators and pathogens, their densities, and thus their effects, became very high (Freeland, 1990).

Similarly in Britain many plants are avoided by grazing animals because they are distasteful, hairy, prickly, or even poisonous (Tivy, 1971). The persistence and continued spread of bracken (*Pteridium aquilinium*) on heavily grazed rough pasture in Scotland is aided by the fact that it is slightly poisonous, especially to young stock. The success of bracken is furthered by its reaction to burning, for with its extensive system of underground stems (rhizomes) it tends to be little damaged by fire. The survival and prevalence of shrubs such as elder (*Sambucus nigra*), gorse (*Ulex* spp.), broom (*Sarothamus scoparius*), and the common weeds ragwort (*Senecio jacobaea*) and creeping thistle (*Cirsium arvense*) in the face of grazing can be attributed to their lack of palatability.

The role of grazing in causing marked deterioration of habitat has been the subject of further discussion in the context of upland Britain. In particular, Darling (1956) stressed that while trees bring up nutrients from rocks and keep minerals in circulation, pastoralism means the export of calcium phosphate and nitrogenous organic matter. The vegetation gradually deteriorates, the calcicoles disappear, and the herbage becomes deficient in both minerals and protein. Progressively more xerophytic plants come in: *Nardus stricta*, *Molinia caerulaea*, *Erica tetralix*, and then *Scripus caespitosa*. However, this is not a view that now receives universal support (Mather, 1983). Studies in several parts of upland Britain have shown that mineral inputs from precipitation are very much greater than the nutrient losses in wool and sheep carcasses. During the 1970s the idea of upland deterioration was gradually undermined.

In general terms it is clear that in many part of the world the grass family is well equipped to withstand grazing. Many plants have their growing points located on the apex of leaves and shoots, but grasses reproduce the bulk of fresh tissue at the base of their leaves. This part is least likely to be damaged by grazing and allows regrowth to continue at the same time that material is being removed.

Communities severely affected by the treading of animals (and indeed people) tend to have certain distinctive characteristics. These include diminutiveness (since the smaller the plant is, the more protection it will get from soil surface irregularities); strong ramification (the plant stems and leaves spread close to the ground); small leaves (which are less easily damaged by treading); tissue firmness (cell-wall strength and thickness to limit mechanical damage); a bending ability; strong vegetative increase and dispersal (e.g., by stolons); small hard seeds which can be easily dispersed, and the production of a large number of seeds per plant (which is particularly important because the mortality of seedlings is high under treading and trampling conditions).

Deforestation

Deforestation is one of the great processes of landscape transformation (Williams, 2003). However, controversy surrounds the meaning of the word 'deforestation' and this causes problems when it comes to assessing rates of change and causes of the phenomenon (Williams, 1994). It is best defined (Grainger, 1992) as 'the temporary or permanent clearance of forest for agriculture or other purposes'. According to this definition, if clearance does not take place then deforestation does not occur. Thus, much logging in the tropics, which is selective in that only a certain proportion of trees and only certain species are removed, does not involve clear felling and so cannot be said to constitute deforestation. However, how 'clear' is clear? Many shifting cultivators in the humid tropics leave a small proportion of trees standing (perhaps because they have special utility). At what point does the proportion of trees left standing permit one to say that deforestation has taken place? This is not a question to which there is a simple answer. There is not even a globally accepted definition of forest, and the UN's Food and Agriculture Organization (FAO) defines forest cover as greater than 10% canopy cover, whereas the International Geosphere Biosphere Program (IGBP) defines it as greater than 60%.

The deliberate removal of forest is one of the most longstanding and significant ways in which humans have modified the environment, whether achieved by fire or cutting. Pollen analysis shows that temperate forests were removed in Mesolithic and Neolithic times and at an accelerating rate thereafter. The example of Easter Island in the Pacific is salutary, for its deforestation at an unsustainable rate caused the demise of the culture that created its great statues. Since pre-agricultural times forests have declined by approximately one-fifth, from five to four billion hectares. The highest loss has occurred in temperate forests (32–5%), followed by subtropical woody savannas and deciduous forests (24–5%). The lowest losses have been in tropical evergreen forests (4–6%) because many of them have for much of their history been inaccessible and sparsely populated (World Resources Institute, 1992: 107).

In Britain, Birks (1988) analyzed dated pollen sequences to ascertain when and where tree pollen values in sediments from upland areas drop to 50% of their Holocene maximum percentages, and takes this as a working definition of 'deforestation'. From this work he identified four phases:

1 *3700–3900 years BP*: northwest Scotland and the eastern Isle of Skye;
2 *2100–2600 years BP (the pre-Roman iron age)*: Wales, England (except the Lake District), northern Skye, and northern Sutherland;
3 *1400–1700 years BP (post-Roman)*: Lake District, southern Skye, Galloway and Knapdale–Ardnamurchan;
4 *300–400 years BP*: the Grampians and the Cairngorms.

Sometimes forests are cleared to allow agriculture; at other times to provide fuel for domestic purposes, or

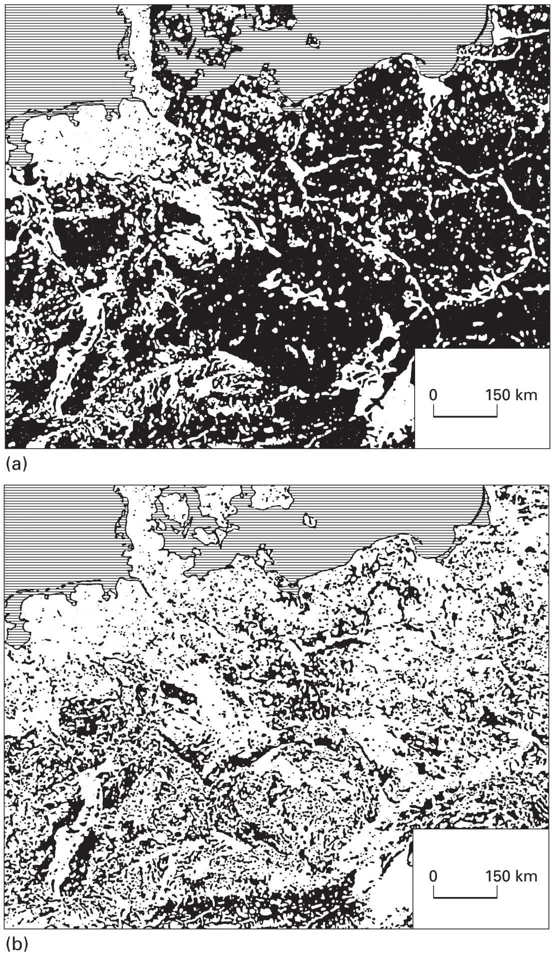

Figure 2.8 The changing distribution of forest in central Europe between (a) AD 900 and (b) AD 1900 (reprinted from Darby, 1956: 202–3, in *Man's role in changing the face of the Earth*, ed. W. L. Thomas, by permission of the University of Chicago Press © The University of Chicago 1956).

to provide charcoal or wood for construction; sometimes to fuel locomotives, or to smoke fish; and sometimes to smelt metals. The Phoenicians were exporting cedars as early as 4600 years ago (Mikesell, 1969), both to the Pharaohs and to Mesopotamia. Attica in Greece was laid bare by the fifth century BC and classical writers allude to the effects of fire, cutting and the destructive nibble of the goat. The great phase of deforestation in central and western Europe, described by Darby (1956: 194) as 'the great heroic period of reclamation', occurred properly from AD 1050 onwards for about 200 years (Figure 2.8). In particular, the Germans moved eastward: 'What the new west meant to young America in the nineteenth century, the new east meant to Germany in the Middle Ages' (Darby, 1956: 196). The landscape of Europe was transformed, just as that of North America, Australia, New Zealand, and South Africa was to be as a result of the European expansions, especially in the nineteenth century.

Temperate North America underwent particularly brutal deforestation (Williams, 1989), and lost more woodland in 200 years than Europe did in 2000. The first colonialists arriving in the *Mayflower* found a continent that was wooded from the Atlantic seaboard as far as the Mississippi River (Figure 2.9). The forest originally occupied some 170 million hectares. Today only about 10 million hectares remain.

Fears have often been expressed that the mountains of High Asia (e.g., in Nepal) have been suffering from a wave of deforestation that has led to a whole suite of environmental consequences, which include accelerated landsliding, flooding in the Ganges Plain and sedimentation in the deltaic areas of Bengal. Ives and Messerli (1989) doubt that this alarmist viewpoint is soundly based and argue (p. 67) that 'the popular claims about catastrophic post-1950 deforestation of the Middle Mountain belt and area of the high mountains of the Himalayas are much exaggerated, if not inaccurate.'

Likewise, some workers in West Africa have interpreted 'islands' of dense forest in the savanna as the relics of a once more extensive forest cover that was being rapidly degraded by human pressure. More recent research (Fairhead and Leach, 1996) suggests that this is far from the truth and that villagers have in recent decades been extending rather than reducing forest islands (Table 2.1). There are also some areas of Kenya in East Africa where, far from recent population pressures promoting vegetation removal and land degradation, there has been an increase in woodland since the 1930s.

With regard to the equatorial rain forests, the researches of Flenley and others (Flenley, 1979) have indicated that forest clearance for agriculture has been going on since at least 3000 years BP in Africa, 7000 years BP in South and Central America, and possibly since 9000 years BP or earlier in India and New Guinea. Recent studies by archeologists and paleoecologists have tended to show that prehistoric human activities were rather more extensive in the tropical forests than originally thought (Willis et al., 2004).

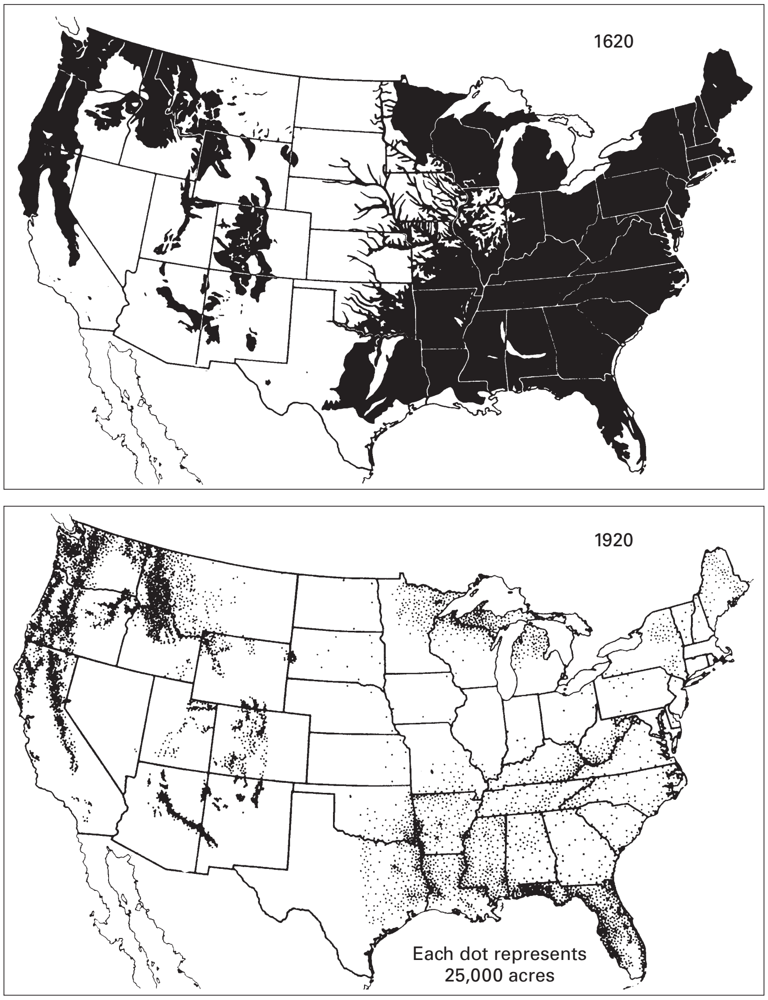

Figure 2.9 The distribution of American natural forest in 1620 and 1920 (modified from Williams, 1989).

Table 2.1 Estimates of deforestation since 1900 (millions of hectares)

Country	*Orthodox estimate of forest area lost*	*Forest area lost according to World Conservation Monitoring Center*	*Forest area lost according to Fairhead and Leach (1996, table 9.1)*
Côte d'Ivoire (Ivory Coast)	13	20.2	4.3–5.3
Liberia	4–4.5	5.5	1.3
Ghana	7	12.9	3.9
Benin	0.7	12.9	3.9
Togo	0	1.7	0
Sierra Leone	0.8–5	6.7	*c.* 0
Total	25.3–30.2	48.6	9.5–10.5

The causes of the present spasm of deforestation in the tropics (Figure 2.10) are complex and multifarious and are summarized in Grainger (1992) and Table 2.2. Grainger also provides an excellent review of the problems of measuring and defining loss of tropical forest area. Indeed, there are very considerable difficulties in estimating rates of deforestation, in part because different groups of workers use different definitions of what constitutes a forest (Allen and Barnes, 1985) and what distinguishes rain forest from other types of forest. There is thus some variability in views as to the present rate of rain forest removal and this is brought out in a recent debate (see Achard et al., 2002; Fearnside and Lawrence, 2003). The FAO estimates (Lanly et al., 1991) that the total annual deforestation in 1990 for 62

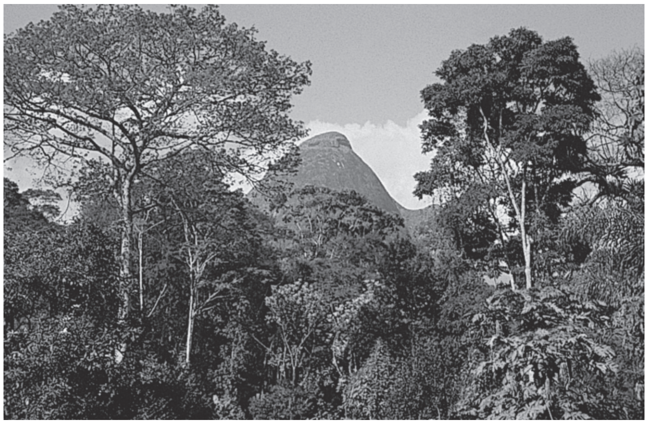

Figure 2.10 One of the most serious environmental problems in tropical areas is the removal of the rain forest. The felling of trees in Brazil on slopes as steep as these will cause accelerated erosion and loss of soil nutrients, and may promote lateritization.

Table 2.2 The causes of deforestation. Source: from Grainger (1992)

Immediate causes – land use	*Underlying causes*
Shifting agriculture:	Socio-economic mechanisms:
(a) traditional long-rotation shifting cultivation	(a) population growth
(b) short-rotation shifting cultivation	(b) economic development
(c) encroaching cultivation	Physical factors:
(d) pastoralism	(a) distribution of forests
Permanent agriculture:	(b) proximity of rivers
(a) permanent staple crop cultivation	(c) proximity of roads
(b) fish farming	(d) distance from urban centers
(c) Government sponsored resettlement schemes	(e) topography
(d) cattle ranching	(f) soil fertility
(e) tree crop and other cash crop plantations	Government policies:
Mining	(a) agriculture policies
Hydroelectric schemes	(b) forestry policies
Cultivation of illegal narcotics	(c) other policies

countries (representing some 78% of the tropical forest area of the world) was 16.8 million hectares, a figure significantly higher than the one obtained for these same countries for the period 1976–80 (9.2 million hectares per year). Myers (1992) suggests that there was an 89% increase in the tropical deforestation rate during the 1980s (compared with an FAO estimate of a 59% increase). He believed that the annual rate of loss in 1991 amounted to about 2% of the total forest expanse.

The change in forest area on a global basis in the 1990s is shown in Table 2.3a. These FAO data indicate the ongoing process of deforestation in Africa, Asia and the Pacific, Latin America and the Caribbean, and the stable or slightly reducing amount of deforestation in Europe, West Asia, and North America.

There is, however, a considerable variation in the rate of forest regression in different areas (Figure 2.11), with some areas under relatively modest threat (e.g., western Amazonia, the forests of Guyana, Surinam and French Guyana, and much of the Zaire Basin in central Africa (Myers, 1983, 1984)). Some other areas are being exploited so fast that minimal areas will soon be left, for example: the Philippines, peninsular Malaya, Thailand, Australia, Indonesia, Vietnam, Bangladesh, Sri Lanka, Central America, Madagascar, West Africa, and eastern Amazonia. Myers (1992) refers to particular 'hot spots' where the rates of deforestation are especially threatening, and presents data for certain locations where the percentage loss of forest is more than three times his global figure of 2%: southern Mexico (10%), Madagascar (10%), northern and eastern Thailand (9.6%), Vietnam (6.6%), and the Philippines (6.7%).

A more recent examination of rate of deforestation in selected hot spots is provided by Achard et al. (2002) (Table 2.3b). They suggest that the annual deforestation rate for rain forests is 0.38% for Latin America, 0.43% for Africa, and 0.91% for Southeast Asia, but that the hot spot areas can have rates that are much greater (e.g., up to 5.9% in central Sumatra and up to 4.7% in Madagascar). These rates are not as high as those of Myers, but are still extremely rapid.

The rapid loss of rain forest is potentially extremely serious, because as Poore (1976: 138) stated, these forests are a source of potential foods, drinks, medicines, contraceptives, abortifacients, gums, resins, scents, colorants, specific pesticides, and so on. Their removal may contribute to crucial global environmental concerns (e.g., climatic change and loss of biodiversity), besides causing regional and local problems, including lateritization, increased rates of erosion, and accelerated mass movements. The great range of potential impacts of tropical deforestation is summarized in Table 2.4.

Table 2.3
(a) Change in forested land 1990–2000 by region. Source: compiled from FAO (2001)

	Total land area (10^6 hectares)	*Total forest 1990 (10^6 hectares)*	*Total forest 2000 (10^6 hectares)*	*Percent of land forested in 2000*	*Change 1990–2000 (10^6 hectares)*	*Percent change per year*
Africa	2963.3	702.5	649.9	21.9	–52.6	–0.7
Asia and the Pacific	3463.2	734.0	726.3	21.0	–7.7	–0.1
Europe	2359.4	1042.0	1051.3	44.6	9.3	0.1
Latin America and the Caribbean	2017.8	1011.0	964.4	47.8	–46.7	–0.5
North America	1838.0	466.7	470.1	25.6	3.9	0.1
West Asia	372.4	3.6	3.7	1.0	0.0	0.0
World*	13,014.1	3960.0	3866.1	29.7	–93.9	–0.24

*Numbers may not add due to rounding.

(b) Annual deforestation rates as a percentage of the 1990 forest cover, for selected areas of rapid forest cover change (hot spots) within each continent. Source: Achard et al. (2002)

Hot-spot areas by continent	*Annual deforestation rate of sample sites within hot-spot area (range), %*
Latin America	0.38
Central America	0.8–1.5
Brazilian Amazonian belt	
Acre	4.4
Rondônia	3.2
Mato Grosso	1.4–2.7
Pará	0.9–2.4
Columbia–Ecuador border	*c.* 1.5
Peruvian Andes	0.5–1.0
Africa	0.43
Madagascar	1.4–4.7
Côte d'Ivoire	1.1–2.9
Southeast Asia	0.91
Southeastern Bangladesh	2.0
Central Myanmar	*c.* 3.0
Central Sumatra	3.2–5.9
Southern Vietnam	1.2–3.2
Southeastern Kalimantan	1.0–2.7

Some traditional societies have developed means of exploiting the rain forest environment which tend to minimize the problems posed by soil fertility deterioration, soil erosion, and vegetation degradation. Such a system is known as shifting agriculture (*swidden*). As Geertz (1963: 16) remarked:

> In ecological terms, the most distinctive positive characteristic of swidden agriculture . . . is that it is integrated into and, when genuinely adaptive, maintains the general structure of the pre-existing natural ecosystem into which it is projected, rather than creating and sustaining one organized along novel lines and displaying novel dynamics.

The tropical rain forest and the swidden plots have certain common characteristics. Both are closed cover systems, in part because in swidden some trees are left standing, in part because some tree crops (such as banana, papaya, areca, etc.) are planted, but also because food plants are not planted in an open field,

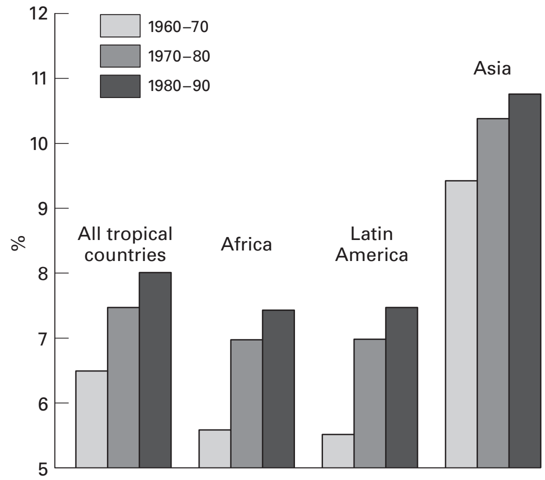

Figure 2.11 Estimated rate of tropical deforestation, 1960–1990, showing the particularly high rates in Asia (modified from *World Resources, 1996–97*, World Resources Institute, 1998).

crop-row manner, but helter-skelter in a tightly woven, dense botanical fabric. It is in Geertz's words (1963: 25) 'a miniaturized tropical forest'. Second, swidden agriculture normally involves a wide range of cultigens, thereby having a high diversity index like the rain forest itself. Third, both swidden plots and the rain forest have high quantities of nutrients locked up in the biotic community (Douglas, 1969) compared to that in the soil. The primary concern of 'slash-and-burn' activities is not merely the clearing of the land, but rather the transfer of the rich store of nutrients locked up in the prolific vegetation of the rain forest to a botanical complex whose yield to people is a great deal larger. If the period of cultivation is not too long and the period of fallow is long enough, an equilibrated, non-deteriorating and reasonable productive farming regime can be sustained in spite of the rather impoverished soil base upon which it rests.

Unfortunately this system often breaks down, especially when population increase precludes the maintenance of an adequately long fallow period. When this happens, the rain forest cannot recuperate and is replaced by a more open vegetation assemblage ('derived savanna') that is often dominated by the notorious *Imperata* savanna grass, which has turned so much of Southeast Asia into a green desert. *Imperata cyclindrica* is a tall grass that springs up from rhizomes. Because of its rhizomes it is fire-resistant, but since it is a tall grass it helps to spread fire (Gourou, 1961).

One particular type of tropical forest ecosystem coming under increasing pressure from various human activities is the mangrove forest (Figure 2.12) characteristic of intertidal zones. These ecosystems constitute a reservoir, refuge, feeding ground, and nursery for many useful and unusual plants and animals (Mercer and Hamilton, 1984). In particular, because they export decomposable plant debris into adjacent coastal waters, they provide an important energy source and nutrient input to many tropical estuaries. In addition they can serve as buffers against the erosion caused by tropical storms – a crucial consideration in low-lying areas such as Bangladesh. In spite of these advantages, mangrove forests are being degraded and destroyed on a large scale in many parts of the world, either through exploitation of their wood resources or because of their conversion to single-use systems such as agriculture, aquaculture, salt-evaporation ponds, or housing developments. Data from FAO suggest that the world's mangrove forests, which covered 19.8 million hectares in 1980, have now been reduced to only 14.7 million hectares, with the annual loss running at about 1% per year (compared with 1.7 % a year from

Table 2.4 The consequences of tropical deforestation. Source: from Grainger (1992)

Reduced biological diversity	*Changes in local and regional environments*	*Changes in global environments*
Species extinction	More soil degradation	Reduction in carbon stored in terrestrial biota
Reduced capacity to breed improved crop varieties	Changes in water flows from catchments	Increase in carbon dioxide content of atmosphere
Inability to make some plants economic crops	Changes in buffering of water flows by wetland forests	Changes in global temperature and rainfall patterns by greenhouse effects
Threat to production of minor forest products	Increased sedimentation of rivers, reservoirs etc.	Other changes in global climate due to changes in land surface processes
	Possible changes in rainfall characteristics	

Figure 2.12 Mangrove swamps are highly productive and diverse ecosystems that are being increasingly abused by human activities.

1980 to 1990 (http://www.fao.org/DOCREP/005/Y7581E/y7581e04.htm).

On a global basis Richards (1991: 164) has calculated that since 1700 about 19% of the world's forests and woodlands have been removed. Over the same period the world's cropland area has increased by over four and a half times, and between 1950 and 1980 it amounted to well over 100,000 km^2 per year.

Deforestation is not an unstoppable or irreversible process. In the USA the forested area has increased substantially since the 1930s and 1940s. This 'rebirth of the forest' (Williams, 1988) has a variety of causes: new timber growth and planting was rendered possible because the old forest had been removed, forest fires have been suppressed and controlled, farmland has been abandoned and reverted to forest, and there has been a falling demand for lumber and lumber-derived products. It is, however, often very difficult to disentangle the relative importance of grazing impacts, fire, and fluctuating climates in causing changes in the structure of forested landscapes. In some cases, as for example the Ponderosa pine forests of the American southwest, all three factors may have contributed to the way in which the park-like forest of the nineteenth century has become significantly denser and younger today (Savage, 1991).

Given the perceived and actual severity of tropical deforestation it is evident that strategies need to be developed to reduce the rate at which it is disappearing. Possible strategies include the following:

- *research, training, and education* to give people a better understanding of how forests work and why they are important, and to change public opinion so that more people appreciate the uses and potential of forests;
- *land reform* to reduce the mounting pressures on landless peasants caused by inequalities in land ownership;
- *conservation of natural resources* by setting aside areas of rain forest as National Parks or nature reserves;
- *restoration and reforestation* of damaged forests;
- *sustainable development*, namely development which, while protecting the habitat, allows a type and level of economic activity that can be sustained into the future with minimum damage to people or forest (e.g., selective logging rather than clear felling; promotion of nontree products; small-scale farming in plots within the forest);
- *control of the timber trade* (e.g., by imposing heavy taxes on imported tropical forest products and outlawing the sale of tropical hardwoods from nonsustainable sources);
- *'debt-for-nature' swaps* whereby debt-ridden tropical countries set a monetary value on their ecological capital assets (in this case forests) and literally trade them for their international financial debt;
- *improvement of local peoples* in managing the remaining rain forests;
- *careful control of international aid* and development funds to make sure that they do not inadvertently lead to forest destruction;
- *reducing demand for wood products.*

Having considered the importance of the three basic processes of fire, grazing, and deforestation, we can now turn to a consideration of some of the major changes in vegetation types that have taken place over extensive areas as a result of such activities.

Secondary rain forest

When an area of rain forest that has been cleared for cultivation or timber exploitation is abandoned by humans, the forest begins to regenerate, but for an extended period of years the type of forest that occurs – **secondary forest** – is very different in character from the virgin forest it replaces. The features of such secondary forest, which is widespread in many tropical regions and accounts for as much as 40% of the total forest area in the tropics, have been summarized by Richards (1952), Ellenberg (1979), Brown and Lugo (1990), and Corlett (1995).

First, secondary forest is lower and consists of trees of smaller average dimensions than those of primary forest, but since it is comparatively rare that an area of primary forest is clear-felled or completely destroyed by fire, occasional trees much larger than average are usually found scattered through secondary forest. Second, very young secondary forest is often remarkably regular and uniform in structure, although the abundance of small climbers and young saplings gives it a dense and tangled appearance, which is unlike that of primary forest and makes it laborious to penetrate. Third, secondary forest tends to be much poorer in species than primary, and is sometimes, although by no means always, dominated by a single species, or a small number of species. Fourth, the dominant trees of secondary forest are light-demanding and intolerant of shade, most of the trees possess efficient dispersal mechanisms (having seeds or fruits well adapted for transport by wind or animals), and most of them can grow very quickly. The rate of net primary production of secondary forests exceeds that of the primary forests by a factor of two (Brown and Lugo, 1990). Some species are known to grow at rates of up to 12 m in 3 years, but they tend to be short-lived and to mature and reproduce early. One consequence of their rapid growth is that their wood often has a soft texture and low density.

Secondary forests should not be dismissed as useless scrub. All but the youngest secondary forests are probably effective at preventing soil erosion, regulating water supply and maintaining water quality. They provide refuge for some flora and fauna and can provide a source of timber, albeit generally of inferior quality than derived from primary forest. In addition, secondary forest regrowth can have a profound effect on water resources (see, e.g., Wilby and Gell, 1994).

The human role in the creation and maintenance of savanna

The **savannas** (Figure 2.13) can be defined, following Hills (1965: 218–19), as:

Figure 2.13 A tropical savanna in the Kimberley District of northwest Australia.

> a plant formation of tropical regions, comprising a virtually continuous ecologically dominant stratum of more or less xeromorphic plants, of which herbaceous plants, especially grasses and sedges, are frequently the principal, and occasionally the only, components, although woody plants often of the dimension of trees or palms generally occur and are present in varying densities.

They are extremely widespread in low latitudes (Harris, 1980), covering about 18 million km^2 (an area about 2.6 million km^2 greater than that of the tropical rain forest). Their origin has been the subject of great contention in the literature of biogeography, although most savanna research workers agree that no matter what the savanna origins may be, the agent that seems to maintain them is intentional or inadvertent burning (Scott, 1977). As with most major vegetation types a large number of interrelated factors are involved in causing savanna (Mistry, 2000), and too many arguments about origins have neglected this fact (Figure 2.14). Confusion has also arisen because of the failure to distinguish clearly between predisposing, causal, resulting, and maintaining factors (Hills, 1965). It appears, for instance, that in the savanna regions around the periphery of the Amazon basin the climate *predisposes* the vegetation towards the development of savanna rather than forest. The geomorphic evolution of the landscape may be a *causal* factor; increased laterite development a *resulting* factor; and fire, *a maintaining* factor.

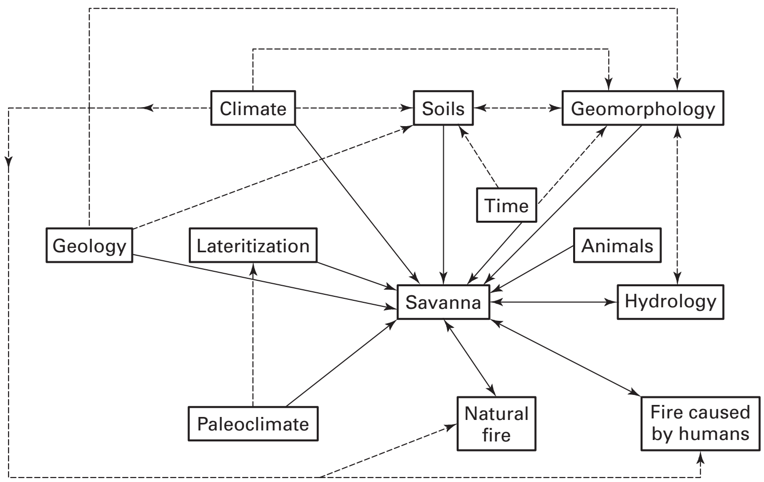

Figure 2.14 The interrelated factors involved in the formation of savanna vegetation.

Originally, however, savanna was envisaged as a predominantly natural vegetation type of climatic origin (see, e.g., Schimper, 1903), and the climatologist Köppen used the term 'savanna climate', implying that a specific type of climate is associated with all areas of savanna origin. According to supporters of this theory, savanna is better adapted than other plant formations to withstand the annual cycle of alternating soil moisture: rain forests could not resist the extended period of extreme drought, whereas dry forests could not compete successfully with perennial grasses during the equally lengthy period of large water surplus (Sarmiento and Monasterio, 1975).

Other workers have championed the importance of edaphic (soil) conditions, including poor drainage, soils which have a low water-retention capacity in the dry season, soils with a shallow profile due to the development of a lateritic crust, and soils with a low nutrient supply (either because they are developed on a poor parent rock such as quartzite, or because the soil has undergone an extended period of leaching on an old land surface). Associated with soil characteristics, ages of land surfaces, and degree of drainage is the geomorphology of an area (as stressed by, e.g., Cole, 1963). This may also be an important factor in savanna development.

Some other researchers – for example, Eden (1974) – found that savannas are the product of former drier conditions (such as late Pleistocene aridity) and that, in spite of a moistening climate, they have been maintained by fire. He pointed to the fact that the patches of savanna in southern Venezuela occur in forest areas of similar humidity and soil infertility, suggesting that neither soil, nor drainage, nor climate was the cause. Moreover, the present 'islands' of savanna are characterized by species which are also present elsewhere in tropical American savannas and whose disjunct distribution conforms to the hypothesis of previous widespread continuity of that formation.

The importance of fire in maintaining and originating some savannas is suggested by the fact that many savanna trees are fire-resistant. Controlled experiments in Africa (Hopkins, 1965) demonstrated that some tree species, such as *Burkea africana* and *Lophira lanceolata*, withstand repeated burning better than others do. There are also many observations of the frequency with which, for example, African herdsmen and agriculturists burn over much of tropical Africa and thereby maintain grassland. Certainly the climate of savanna areas is conducive to fire for, as Gillian (1983: 617) put it, 'Large scale grass fires are more likely to take place in areas having a climate moist enough to permit the production of a large amount of grass, but seasonally dry enough to allow the dried material to catch fire and burn easily.' On the other hand Morgan and Moss (1965) express some doubts about the role of fire in western Nigeria, and point out that fire is not itself necessarily an independent variable:

> The evidence suggests that some notions of the extent and destructiveness of savanna fires are rather exaggerated. In particular the idea of an annual burn that affects a large

> proportion of the area in each year could seem to be false. It is more likely that some patches, peculiarly susceptible to fire, as a result of edaphic or biotic influences upon the character of the community itself, are repeatedly burned, whereas others are hardly, if ever, affected . . . It is also important to note that there is no evidence anywhere along the forest fringe . . . to suggest that fire sweeps into the forest, effecting notable destruction of forest trees.

Some savannas are undoubtedly natural, for pollen analysis in South America shows that savanna vegetation was present before the arrival of human civilization. Nonetheless even natural savannas, when subjected to human pressures, change their characteristics. For example, the inability of grass cover to maintain itself over long periods in the presence of heavy stock grazing may be documented from many of the warm countries of the world (Johannessen, 1963). Heavy grazing tends to remove the fuel (grass) from much of the surface. The frequency of fires is therefore significantly reduced, and tree and bush invasion take place. As Johannessen (p. 111) wrote: 'Without intense, almost annual fires, seedlings of trees and shrubs are able to invade the savannas where the grass sod has been opened by heavy grazing . . . the age and size of the trees on the savannas usually confirm the relative recency of the invasion.' In the case of the savannas of interior Honduras, he reports that they only had a scattering of trees when the Spaniards first encountered them, whereas now they have been invaded by an assortment of trees and tall shrubs.

Elsewhere in Latin America changes in the nature and density of population have also led to a change in the nature and distribution of savanna grassland. C. F. Bennett (1968: 101) noted that in Panama the decimation of the Indian population saw the re-establishment of trees:

> Large areas that today are covered by dense forest were in farms, grassland or low second growth in the early sixteenth century when the Spaniards arrived. At that time horses were ridden with ease through areas which today are most easily penetrated by river, so dense has the tree growth become since the Indians died away.

Such bush or shrub encroachment is a serious cause of rangeland deterioration in savanna regions (Rogues et al., 2001). Overgrazing reduces the vigor of favorable perennial grasses, which tend to be replaced, as we have seen, by less reliable annuals and by woody vegetation. Annual grasses do not adequately hold or cover the soil, especially early in the growing season. Thus runoff increases and topsoil erosion occurs. Less water is then available in the topsoil to feed the grasses, so that the woody species, which depend on deeper water, become more competitive relative to the grasses, which can use water only within their shallow root zones. The situation is exacerbated when there are few browsers in the herbivore population and, once established, woody vegetation competes effectively for light and nutrients. It is also extremely expensive to remove established dense scrub cover by mechanical means, although some success has been achieved by introducing animals that have a browsing or bulldozing effect, such as goats, giraffes, and elephants. Experiments in Zimbabwe have shown that, if bush clearance is carried out, a threefold increase in the sustainable carrying capacity can take place in semi-arid areas (Child, 1985).

Another example of human interventions modifying savanna character is through their effects on savanna-dwelling mammals such as the elephant. They are what is known as a '**keystone species**' because they exert a strong influence on many aspects of the environment in which they live (Waithaka, 1996). They diversify the ecosystems that they occupy and create a mosaic of habitats by browsing, trampling, and knocking over of bushes and trees. They also disperse seeds through their eating and defecating habits and maintain or create water holes by wallowing. All these roles are beneficial to other species. Conversely, where human interference prevents elephants from moving freely within their habitats and leads to their numbers exceeding the carrying capacity of the savanna, their effect can be environmentally catastrophic. Equally, if humans reduce elephant numbers in a particular piece of savanna, the savanna may become less diverse and less open, and its water holes may silt up. This will be to the detriment of other species.

In addition to its lowland savannas, Africa has a series of high-altitude grasslands called 'Afromontane grasslands'. They extend as a series of 'islands' from the mountains of Ethiopia to those of the Cape area of South Africa. Are they the result mainly of forest clearance by humans in the recent past? Or are they a long-standing and probably natural component of the pattern of vegetation (Meadows and Linder, 1993)?

Are they caused by frost, seasonal aridity, excessively poor soils, or an intensive fire regime? This is one of the great controversies of African vegetation studies.

Almost certainly a combination of factors has given rise to these grasslands. On the one hand current land management practices, including the use of fire, prevent forest from expanding. There has undoubtedly been extensive deforestation in recent centuries. On the other hand, pollen analysis from various sites in southern Africa suggests that grassland was present in the area as long ago as 12,000 years BP. This would mean that much grassland is not derived from forest through very recent human activities.

The spread of desert vegetation on desert margins

One of the most contentious and important environmental issues of recent years has been the debate on the question of the alleged expansion of deserts (Middleton and Thomas, 1997).

The term 'desertification' was first used but not formally defined by Aubréville (1949), and for some years the term 'desertization' was also employed, for example by Rapp (1974: 3) who defined it as: 'The spread of desert-like conditions in arid or semi-arid areas, due to man's influence or to climatic change'.

There has been some variability in how 'desertification' itself is defined. Some definitions stress the importance of human causes (e.g., Dregne, 1986: 6–7):

> Desertification is the impoverishment of terrestrial ecosystems under the impact of man. It is the process of deterioration in these ecosystems that can be measured by reduced productivity of desirable plants, undesirable alterations in the biomass and the diversity of the micro and macro fauna and flora, accelerated soil deterioration, and increased hazards for human occupancy.

Others admit the possible importance of climatic controls but give them a relatively inferior role (e.g., Sabadell et al., 1982: 7):

> The sustained decline and/or destruction of the biological productivity of arid and semi arid lands caused by man made stresses, sometimes in conjunction with natural extreme events. Such stresses, if continued or unchecked, over the long term may lead to ecological degradation and ultimately to desert-like conditions.

Yet others, more sensibly, are even-handed or open-minded with respect to natural causes (e.g., Warren and Maizels, 1976: 1):

> A simple and graphic meaning of the word 'desertification' is the development of desert like landscapes in areas that were once green. Its practical meaning . . . is a sustained decline in the yield of useful crops from a dry area accompanying certain kinds of environmental change, both natural and induced.

It is also by no means clear how extensive desertification is or how fast it is proceeding. Indeed, the lack of agreement on the former makes it impossible to determine the latter. As Grainger (1990: 145) has remarked in a well-balanced review, 'Desertification will remain an ephemeral concept to many people until better estimates of its extent and rate of increase can be made on the basis of actual measurements.' He continues (p. 157): 'The subjective judgements of a few experts are insufficient evidence for such a major component of global environmental change . . . Monitoring desertification in the drylands is much more difficult than monitoring deforestation in the humid tropics, but it should not be beyond the ingenuity of scientists to devise appropriate instruments and procedures.'

The United Nations Environment Program (UNEP) has played a central role in the promotion of desertification as a major environmental issue, as is made evident by the following remark by Tolba and El-Kholy (1992: 134):

> Desertification is the main environmental problem of arid lands, which occupy more than 40% of the total global land area. At present, desertification threatens about 3.6 billion hectares – 70% of potentially drylands, or nearly one-quarter of the total land area of the world. These figures exclude natural hyper-arid deserts. About one sixth of the world's population is affected.

However, in their book *Desertification: exploding the myth*, Thomas and Middleton (1994) have discussed UNEP's views on the amount of land that is desertified. They state:

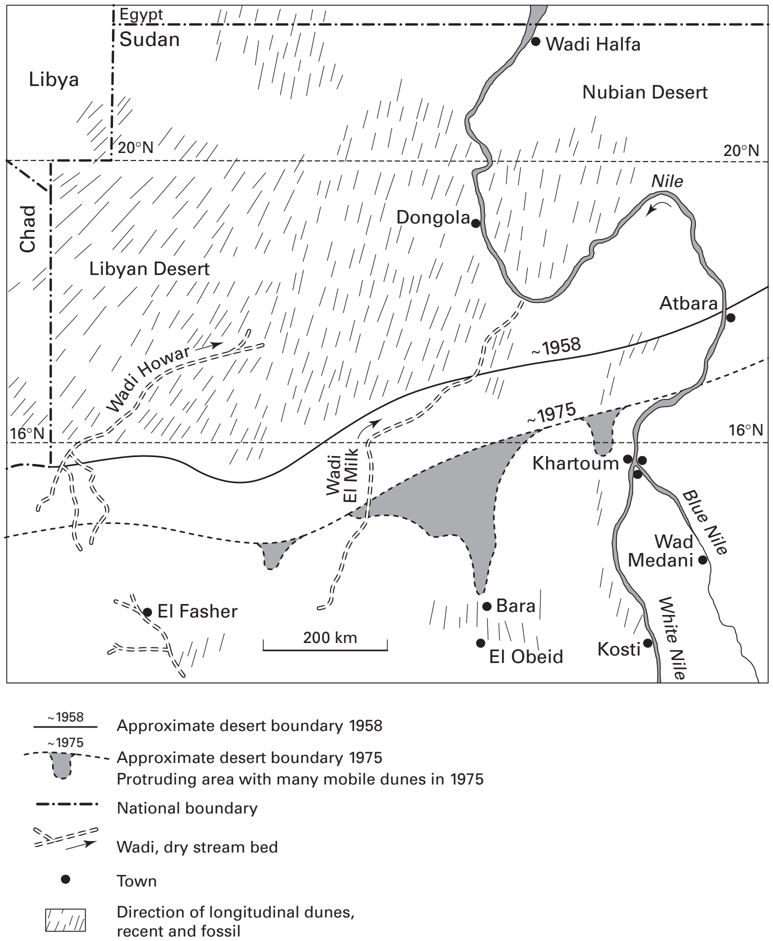

Figure 2.15 Desert encroachment in the northern Sudan 1958–1975, as represented by the position of the boundary between subdesert scrub and grassland in the desert (after Rapp et al., 1976, figure 8.5.3).

> The bases for such data are at best inaccurate and at worst centred on nothing better than guesswork. The advancing desert concept may have been useful as a publicity tool but it is not one that represents the real nature of desertification processes. (Thomas and Middleton, 1994: 160)

Their views have been trenchantly questioned by Stiles (1995), but huge uncertainties do indeed exist.

There are relatively few reliable studies of the rate of supposed desert advance. Lamprey (1975) attempted to measure the shift of vegetation zones in the Sudan (see Figure 2.15) and concluded that the Sahara had advanced by 90 to 100 km between 1958 and 1975, an average rate of about 5.5 km per year. However, on the basis of analysis of remotely sensed data and ground observation, Helldén (1985) found sparse evidence that this had in fact happened. One problem is that there may be very substantial fluctuations in biomass production from year to year. This has been revealed by meteorological satellite observations of green biomass production levels on the south side of the Sahara (Dregne and Tucker, 1988).

The spatial character of desertification is also the subject of some controversy (Helldén, 1985). Contrary to popular rumor, the spread of desert-like conditions is not an advance over a broad front in the way that a wave overwhelms a beach. Rather, it is like a ‘rash’ which tends to be localized around settlements (Figure 2.16). It has been likened to Dhobi’s itch – a ticklish problem in difficult places. Fundamentally, as Mabbutt (1985: 2) has explained, ‘the extension of desert-like conditions tends to be achieved through a process of accretion from without, rather than through expansionary forces acting from within the deserts.’ This distinction is important in that it influences perceptions of appropriate remedial or combative strategies, which are discussed in Goudie (1990).

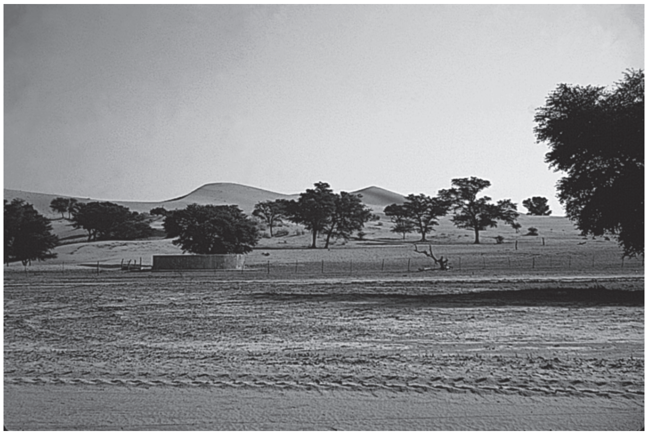

Figure 2.16 On desert margins human activities such as wood collection, overgrazing and cultivation in drought-prone areas have led to the process of desertification. Here we see the effects of overgrazing around a water hole in the Molopo Valley region of the Kalahari, southern Africa.

Figure 2.17 shows two models of the spatial pattern of desertification. One represents the myth of a desert spreading outward over a wide front, whereas the other shows the more realistic pattern of degradation around routes and settlements.

Woodcutting is an extremely serious cause of vegetation decline around almost all towns and cities of the Sahelian and Sudanian zones of Africa. Many people depend on wood for domestic uses (cooking, heating, brick manufacture, etc.), and the collection of wood for charcoal and firewood is an especially serious problem in the vicinity of urban centers. This is illustrated for Khartoum in Sudan in Figure 2.18. Likewise, the installation of modern boreholes has enabled rapid multiplication of livestock numbers and large-scale destruction of the vegetation in a radius of 15–30 km around boreholes (Figure 2.19). Given this localization of degradation, amelioration schemes such as local tree-planting may be partially effective, but ideas of planting green belts as a 'cordon sanitaire' along the desert edge (whatever that is) would not halt deterioration of the situation beyond this Maginot line (Warren and Maizels, 1977: 222). The deserts are not invading from without; the land is deteriorating from within.

There has been considerable debate as to whether the vegetation change and environmental degradation associated with desertization is irreversible. In many cases, where ecological conditions are favorable because of the existence of such factors as deep sandy

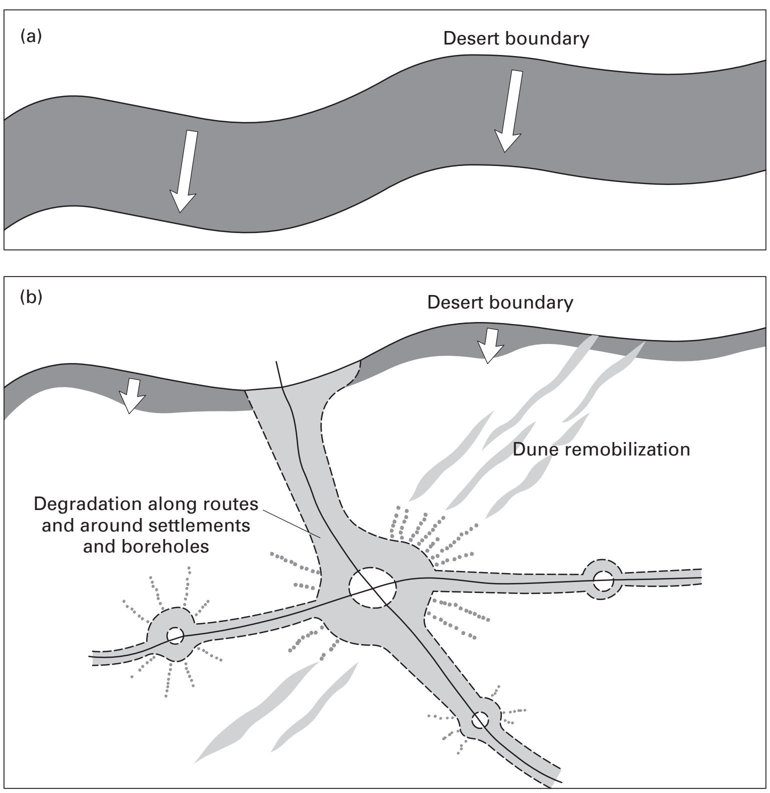

Figure 2.17 Two models of the spatial pattern of desertification: (a) the myth of desert migration over a wide front; (b) the more local development of degradation around routes, settlements, and boreholes (modified after Kadomura, 1994, figure 9).

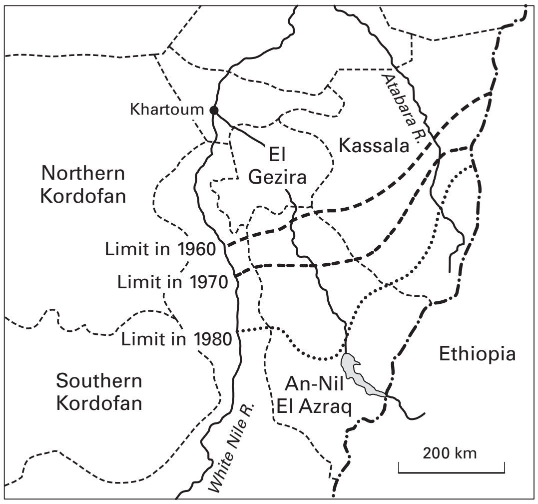

Figure 2.18 The expanding wood and charcoal exploitation zone south of Khartoum, Sudan (after Johnson and Lewis, 1995, figure 6.2).

Figure 2.20 A lichen field, near Swakopmund in Namibia, shows the effects of vehicular traffic. Such scars can take a long time to recover.

soils or beneficial hydrological characteristics, vegetation recovers once excess pressures are eliminated. There is evidence of this in arid zones throughout the world where temporary or permanent enclosures have been set up (Le Houérou, 1977). The speed of recovery will depend on how advanced deterioration is, the size of the area that is degraded, the nature of the soils and moisture resources, and the character of local vegetation. It needs to be remembered in this context that much desert vegetation is adapted to drought and to harsh conditions, and that it often has inbuilt adaptations which enable a rapid response to improved circumstances.

Nonetheless, in certain specific circumstances recovery is slow and so limited that it may be appropriate to talk of 'irreversible desertization'. Le Houérou (1977: 419), for example, has pointed to such a case in North Africa, where, decades after the end of hostilities, wheel tracks and degraded vegetation produced in World War II were still present in the desert. Some desert surfaces, such as those covered by lichens, are easily destroyed by vehicular traffic but take a long time to recover (Figure 2.20).

The causes of desertification are also highly controversial and diverse (Table 2.5). The question has been asked whether this process is the result of temporary drought periods of high magnitude, whether it is due to long-term climatic change towards aridity (either as alleged post-glacial progressive desiccation or as part of a 200-year cycle), whether it is caused by anthropogenic climatic change, or whether it is the result of human action degrading the biological environments in arid zones. There is little doubt that severe droughts do take place (Nicholson, 1978), and that their effects become worse as human and domestic animal populations increase. The devastating drought in the African Sahel from 1968 to 1984 caused greater

Figure 2.19 Relation between spacing of wells and overgrazing: (a) original situation. End of dry season 1. Herd size limited by dry season pasture. Small population can live on pastoral economy. (b) After well-digging but no change in traditional herding. End of dry season 2. Larger total subsistence herd and more people can live there until a situation when grazing is finished in a drought year; then the system collapses (after Rapp, 1974).

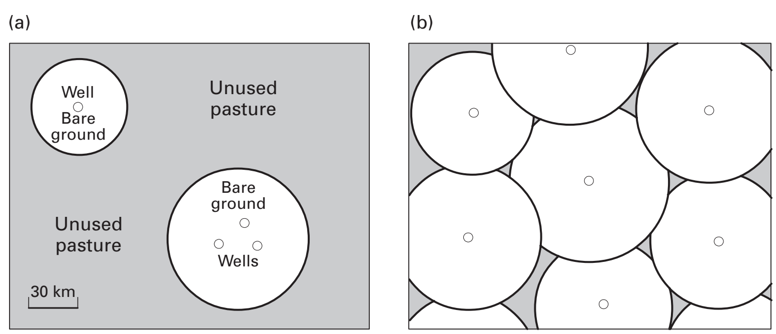

Table 2.5 Some examples of causes and consequences of desertification. Source: Williams (2000, table 2)

Trigger factor	*Consequences*
Direct land use:	Physical processes affected:
(a) overcultivation (decreased fallows, mechanized farming)	(a) decline in soil structure, water permeability, depletion of soil nutrients and organic matter, increased susceptibility to erosion, compaction of soil, sand dune mobilization
(b) overgrazing	(b) loss of biodiversity and biomass, increased soil erosion via wind and water, soil compaction from trampling, increased runoff, sand dune mobilization
(c) mismanagement of irrigated lands	(c) causes water logging and salinization of soil, hence lower crop yields, possible sedimentation of water reservoirs
(d) deforestation (burning, fuel and fodder collection)	(d) promotes artificial establishment of savanna vegetation, loss of soil-stabilizing vegetation, soil exposed and eroded, increases aridity, increased frequency of dust storms, sand dune mobilization (e.g., Ethiopia is losing *c.* 1000 million tonnes of topsoil per year)
(e) exclusion of fire	(e) promotes growth of unpalatable woody shrubs at the expense of herbage
Indirect government policies:	Drives overexploitation land-use practices:
(a) failed population planning policies	(a) increases need for food cultivation, hence overexploitation
(b) irrigation subsidies	(b) exacerbates flooding and salinization
(c) settlement policies/land tenure	(c) forces settlement of nomads, promotes concentrated use of land which often exceeds the carrying capacity
(d) improved infrastructure (e.g., roads, large-scale dams, canals, boreholes)	(d) although beneficial, can exacerbate the problem by attracting increased livestock and human populations or increasing risk from salinization, possible lowering of groundwater table below dams, problems of silting up reservoirs, water-logging; promotes large-scale commercial activity with little local benefit, flooding may displace people and perpetuate cycles of poverty
(e) promotion of cash crops and push towards national/international markets	(e) displaces subsistence cropping, pushes locals into marginal areas to survive, promotes less resilient monocultures, and promotes expansion and intensification of land use
(f) price increases on agricultural produce	(f) incentive to crop on marginal lands
(g) war	(g) valuable resources, both human and financial, are expended on war at the expense of environmental management and the needs of the people, large-scale migration with resultant increased pressure on receiving areas
(h) high interest rates	(h) forcing grazing or cultivation to levels beyond land capacity
Natural:	
(a) extreme drought	(a) decreased vegetation cover and increased land vulnerability for soil erosion. Creates an environment which exacerbates overexploitation
(b) ecological fragility	(b) impact of land-use practices – impact also depends on resilience of environment

ecological stress than the broadly comparable droughts of 1910–15 and 1944–8, largely because of the increasing anthropogenic pressures.

The venerable idea that climate is deteriorating through the mechanism of post-glacial progressive desiccation is now discredited (see Goudie, 1972a, for a critical analysis), although the idea that the Sahel zone is currently going through a 200-year cycle of drought has been proposed (Winstanley, 1973). However, numerous studies of available meteorological data (which in some cases date back as far as 130–150 years) do not allow any conclusions to be reached on the question of systematic long-term changes in rainfall, and the case for climatic deterioration – whether natural or aggravated by humans – is not proven. Indeed, in a judicious review, Rapp (1974: 29) wrote that after consideration of the evidence for the role of climatic change in desertization his conclusion was 'that the reported desertization northwards and southwards from the Sahara could not be explained by a general trend towards drier climate during this century'.

It is evident, therefore, that it is largely a combination of human activities (Figure 2.21) with occasional series of dry years that leads to presently observed desertization. The process also seems to be fiercest not in desert interiors, but on the less arid marginal areas around them. It is in semi-arid areas – where biological productivity is much greater than in extremely

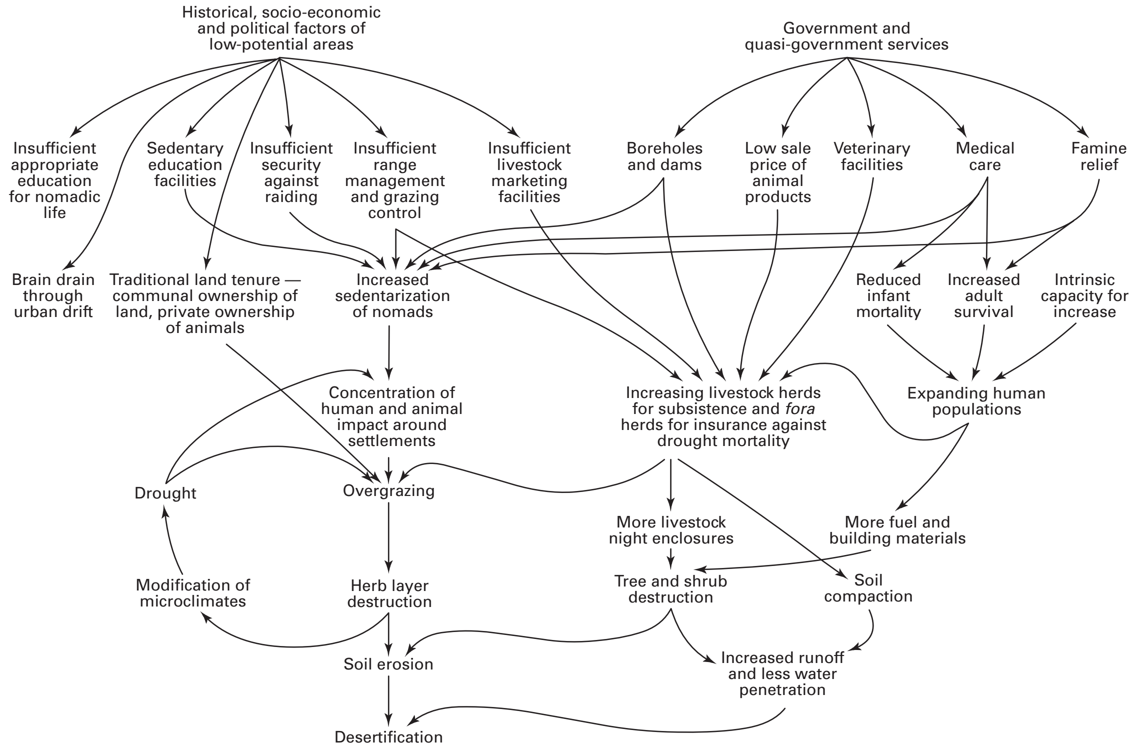

Figure 2.21 Some causal factors in desert encroachment in northern Kenya (after Lamprey, 1975, figure 2).

arid zones, where precipitation is frequent and intense enough to cause rapid erosion of unprotected soils, and where humans are prone to mistake short-term economic gains under temporarily favorable climatic conditions for long-term stability – that the combination of circumstances particularly conducive to desert expansion can be found. It is in these marginal areas that dry farming and cattle rearing can be a success in good years, so that susceptible areas are plowed and cattle numbers become greater than the vegetation can support in dry years. In this way, a depletion of vegetation occurs which sets in train such insidious processes as water erosion and deflation. The vegetation is removed by clearance for cultivation, by the cutting and uprooting of woody species for fuel, by overgrazing and by the burning of vegetation for pasture and charcoal.

These tendencies towards bad land-use practices result in part from the restrictions imposed on many nomadic societies through the imposition of national boundaries across their traditional migration routes, or through various schemes implemented for political and social reasons to encourage their establishment in settled communities. Some of their traditional grazing lands have been taken over by cash-crop farmers. In Niger, for example, there was a sixfold increase in the acreage of peanuts grown between 1934 and 1968. The traditional ability to migrate enabled pastoral nomads and their cattle to emulate the natural migration of such wild animals as wildebeest and kob, and thereby to make flexible use of available resources according to season and according to yearly variations in rainfall. They could also move away from regions that had become exhausted after a long period of use. As soon as migrations are stopped and settlements imposed, such options are closed, and severe degradation occurs (Sinclair and Fryxell, 1985).

The suggestion has sometimes been made that not only are deserts expanding because of human activity, but that deserts themselves are created by human activity. There are authors who have suggested, for example, that the Thar Desert of India is a post-glacial and possibly post-medieval creation (see Allchin et al., 1977, for a critique of such views), while Ehrlich and Ehrlich (1970) have written: ‘The vast Sahara desert itself is largely man-made, the result of over-grazing, faulty irrigation, deforestation, perhaps combined with a shift in the course of a jet stream.' Nothing could be further from the truth. The Sahara, while it has fluctuated greatly in extent, is many millions of years old, pre-dates human life, and is the product of the nature of the general atmospheric circulation, occupying an area of dry descending air.

The maquis of the Mediterranean lands

Around much of the Mediterranean basin there is a plant formation called **maquis** (Figure 2.22). This consists of a stand of xerophilous nondeciduous bushes and shrubs which are evergreen and thick, and whose trucks are normally obscured by low-level branches. It includes such plants as holly oak (*Quercus ilex*), kermes oak (*Quercus coccifera*), tree heath (*Erica arborea*), broom heath (*Erica scoparia*), and strawberry trees (*Arbutus unedo*).

Some of the maquis may represent a stage in the evolution towards true forest in places where the **climax** has not yet been reached, but in large areas it represents the degeneration of forest. Considerable concern has been expressed about the speed with which degeneration to, and degeneration beyond, maquis is taking place as a result of human influences (Tomaselli, 1977), of which cutting, grazing, and fire are probably the most important and long continued. Charcoal burners, goats, and frequent outbreaks of fires among the resinous plants in the dry Mediterranean summer have all taken their toll. Such aspects of degradation in Mediterranean environments are discussed in Conacher and Sala (1998).

Figure 2.22 The maquis of the Mediterranean lands, illustrated here near Marseilles, south of France, is a vegetation type in which humans have played a major role. It represents the degeneration of the natural forest cover.

On the other hand, in some areas, particularly marginal mountainous and semi-arid portions of the Mediterranean basin, agricultural uses of the land have declined in recent decades, as local people have sought easier and more remunerative employment. In such areas scrubland, sometimes termed 'post-cultural shrub formations', may start to invade areas of former cultivation (May, 1991), while maquis has developed to become true woodland. Mediterranean vegetation appears to be very resilient (Grove and Rackham, 2001).

There is considerable evidence that maquis vegetation is in part adapted to, and in part a response to, fire. One effect of fire is to reduce the frequency of standard trees and to favor species that after burning send up a series of suckers from ground level. Both *Quercus ilex* and *Quercus coccifera* seem to respond in this way. Similarly, a number of species (e.g., *Cistus albidus, Erica arborea*, and *Pinus halepensis*) seem to be distinctly advantaged by fire, perhaps because it suppresses competition or perhaps because (as with the comparable chaparral of the southwest USA) a short burst of heat encourages germination (Wright and Wanstall, 1977).

The prairie problem

The mid-latitude grasslands of North America – the prairies – are another major vegetation type that can be used to examine the human impact, although, as in the case of savanna grasslands in the tropics, the human role is the subject of controversy (Whitney, 1994).

It was once fairly widely believed that the prairies were essentially a climatically related phenomenon (see Changuon et al., 2002, for a historical review). Workers such as J. E. Weaver (1954) argued that under the prevailing conditions of soil and climate the invasion and establishment of trees was significantly hindered by the presence of a dense sod. High evapotranspiration levels combined with low precipitation were thought to give a competitive advantage to herbaceous plants with shallow, densely ramifying root systems, capable of completing their life cycles rapidly under conditions of pervasive drought.

An alternative view was, however, put forward by Stewart (1956: 128): 'The fact that throughout the tall-grass prairie planted groves of many species have flourished and have reproduced seedlings during moist years and, furthermore, have survived the most severe and prolonged period of drought in the 1930s suggests that there is no climatic barrier to forests in the area.'

Other arguments along the same lines have been advanced. Wells (1965), for example, has pointed out that in the Great Plains a number of woodland species, notably the junipers, are remarkably drought-resistant, and that their present range extends into the Chihuahua Desert where they often grow in association with one of the most xerophytic shrubs of the American deserts, the creosote bush (*Larrea divaricata*). He remarks (p. 247): 'There is no range of climate in the vast grassland climate of the central plains of North America which can be described as too arid for all species of trees native to the region.' Moreover, confirming Stewart, he points out that numerous plantations and shelterbelts have indicated that trees can survive for at least 50 years in a 'grassland' climate. One of his most persuasive arguments is that, in the distribution of vegetation types in the plains, a particularly striking vegetation feature is the widespread but local occurrence of woodlands along escarpments and other abrupt breaks in topography remote from fluvial irrigation. A probable explanation for this is the fact that fire effects are greatest on flat, level surfaces, where there are high wind speeds and no interruptions to the course of fire. It has also been noted that where burning has been restricted there has been extension of woodland into grassland.

The reasons why fire tends to promote the establishment of grassland have been summarized by Cooper (1961: 150–1).

> In open country fire favours grass over shrubs. Grasses are better adapted to withstand fire than are woody plants. The growing point of dormant grasses from which issues the following years growth, lies near or beneath the ground, protected from all but the severest heat. A grass fire removes only one year's growth, and usually much of this is dried and dead. The living tissue of shrubs, on the other hand, stands well above the ground, fully exposed to fire. When it is burned, the growth of several years is destroyed. Even though many shrubs sprout vigorously after burning, repeated loss of their top growth keeps them small. Perennial grasses, moreover, produce seeds in abundance one or two years after germination; most woody plants require several years to reach seed-bearing age. Fires that are frequent enough to inhibit seed production in woody plants usually restrict the shrubs to a relatively minor part of the grassland area.

Thus, as with savanna, anthropogenic fires may be a factor which maintains, and possibly forms, the prairies and there is now some paleoecological information to support this view (Boyd, 2002). Though again, following the analogy with savanna, it is possible that some of the American prairies may have developed in a post-glacial dry phase, and that with a later increase in rainfall re-establishment of forest cover was impeded by humans through their use of fire and by grazing animals. The grazing animals concerned were not necessarily domesticated, however, for Larson (1940) has suggested that some of the short-grass plains were maintained by wild bison. These, he believed, stocked the plains to capacity so that the introduction of domestic livestock, such as cattle, after the destruction of the wild game was merely a substitution so far as the effect of grazing on plants is concerned. There is indeed pollen analytical evidence that shows the presence of prairie in the western mid-west over 11,000 years ago, prior to the arrival of human settlers (Bernabo and Webb, 1977). Therefore some of it, at least, may be natural.

Comparable arguments have attended the origins of the great Pampa grassland of Argentina. The first Europeans who penetrated the landscape were much impressed by the treeless open country and it was always taken for granted, and indeed became dogma, that the grassland was a primary climax unit. However, this interpretation was successfully challenged, notably by Schmieder (1927a, b) who pointed out that planted trees thrived, that precipitation levels were quite adequate to maintain tree growth and that, in topographically favorable locations such as the steep gullies (*barrancas*) near Buenos Aires, there were numerous endemic representatives of the former forest cover (*monte*). Schmieder believed that Pampa grasslands were produced by a pre-Spanish aboriginal hunting and pastoral population, the density of which had been underestimated, but whose efficiency in the use of fire was proven.

In recent centuries mid-latitude grasslands have been especially rapidly modified by human activities. As Whitney (1994: 257) points out, they have been altered in ways that can be summarized in three phrases: 'plowed out', 'grazed out', and 'worn out'. Very few areas of natural North America prairies remain. The destruction of 'these masterpieces of nature' took less than a century. The preservation of remaining areas of North American grasslands is a major challenge for the conservation movement (Joern and Keeler, 1995). On a global basis it has been estimated that grasslands have been reduced by about 20% from their pre-agricultural extent (Graetz, 1994).

Post-glacial vegetation change in Britain and Europe

The classic interpretation of the vegetational changes of post-glacial (Holocene) times – that is, over the past 11,000 or so years – has been in terms of climate. That changes in the vegetation of Britain and other parts of western Europe took place was identified by pollen analysts and paleobotanists, and these changes were used to construct a model of climatic change – the Blytt–Sernander model. There was thus something of a chicken-and-egg situation: vegetational evidence was used to reconstruct past climates, and past climates were used to explain vegetational change. More recently, however, the importance of humanly induced vegetation changes in the Holocene has been described thus by Behre (1986: vii):

> Human impact has been the most important factor affecting vegetation change, at least in Europe, during the last 7000 years. With the onset of agriculture, at the so-called Neolithic revolution, the human role changed from that of a passive component to an active element that impinged directly on nature. This change had dramatic consequences for the natural environment and landscape development. Arable and pastoral farming, the actual settlements themselves and the consequent changes in the economy significantly altered the natural vegetation and created the cultural landscape with its many different and varying aspects.

Recent paleoecological research has indicated that certain features of the post-glacial pollen record in Europe and Britain can perhaps be attributed to the action of Mesolithic and Neolithic peoples. It is, for example, possible that the expansion of the alder (*Alnus glutinosa*) was not so much a consequence of supposed wetness in the Atlantic period as a result of Mesolithic colonizers. Their removal of natural forest cover helped the spread of alder by reducing competition, as possibly did the burning of reed swamp. It may also have been assisted in its spread by the increased runoff of surface waters occasioned by deforestation and burning of catchment areas. Furthermore, the felling of alder itself promotes vegetative sprouting and cloning, which could result in its rapid spread in swamp forest areas (Moore, 1986).

In the Yorkshire Wolds of northern England pollen analysis suggests that Mesolithic peoples may have caused forest disturbance as early as 8900 years BP. They may even have so suppressed forest growth that they permitted the relatively open landscapes of the early post-glacial to persist as grasslands even when climatic conditions favored forest growth (Bush, 1988).

One vegetational change that has occasioned particular interest is the fall in *Ulmus* pollen – the so-called elm decline – which appears in all pollen diagrams from northwest Europe, though not from North America. A very considerable number of radiocarbon dates have shown that this decline was approximately synchronous over wide areas and took place at about 5000 yr BP (Pennington, 1974). It is now recognized that various hypotheses can be advanced to explain this major event in vegetation history: the original climatic interpretation; progressive soil deterioration; the spread of disease; the role of people.

The climatic interpretation of the elm decline as being caused by cold, wet conditions has been criticized on various grounds (Rackham, 1980: 265):

> A deterioration of climate is inadequate to explain so sudden, universal and specific a change. Had the climate become less favourable for elm, this would not have caused a general decline in elm and elm alone; it would have wiped out elm in areas where the climate had been marginal for it, but would not have affected elm at the middle of its climatic range unless the change was so great as to affect other species also. A climatic change universal in Europe ought to have some effect on North American elms.

Humans may have contributed to soil deterioration and affected elms thereby. Troels-Smith (1956), however, postulated that around 5000 years ago a new technique of keeping stalled domestic animals was introduced by Neolithic peoples, and that these animals were fed by repeated gathering of heavy branches from those trees known to be nutritious – elms. This, it was held, reduced enormously the pollen production of the elms. In Denmark it was found that the first appearance of the pollen of a weed, Ribwort plaintain (*Plantago lanceolata*), always coincided with the fall in elm pollen levels, confirming the association with human settlements.

Experiments have also shown that Neolithic peoples, equipped with polished stone axes, could cut down mature trees and clear by burning a fair-sized patch of established forest within about a week. Such clearings were used for cereal cultivation. In Denmark a genuine chert Neolithic axe was fitted into an ashwood shaft (Figure 2.23). Three men managed to clear about 600 m^2 of birch forest in four hours. Remarkably, more than 100 trees were felled with one axe head, which had not been sharpened for about 4000 years (Cole, 1970: 38). Rackham, however, doubts whether humans alone could have achieved the sheer extent of change in such a short period (see also Peglar and Birks, 1993), and postulates that epidemics of elm disease may have played a role, aided by the fact that the cause of the disease, a fungus called *Ceratocystis*, is particularly attracted to pollarded elms (Rackham, 1980: 266).

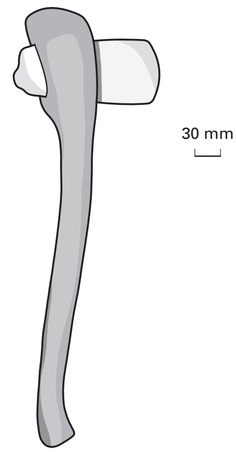

Figure 2.23 A Neolithic chert axe-blade from Denmark, of the type which has been shown to be effective at cutting forest in experimental studies (after Cole, 1970, figure 25).

In reality the elm decline may have resulted from a complex cocktail of causes relating to climate change, soil deterioration, disease and human activity. A model of the interacting factors that could be involved (Parker et al., 2002) is shown in Figure 2.24.

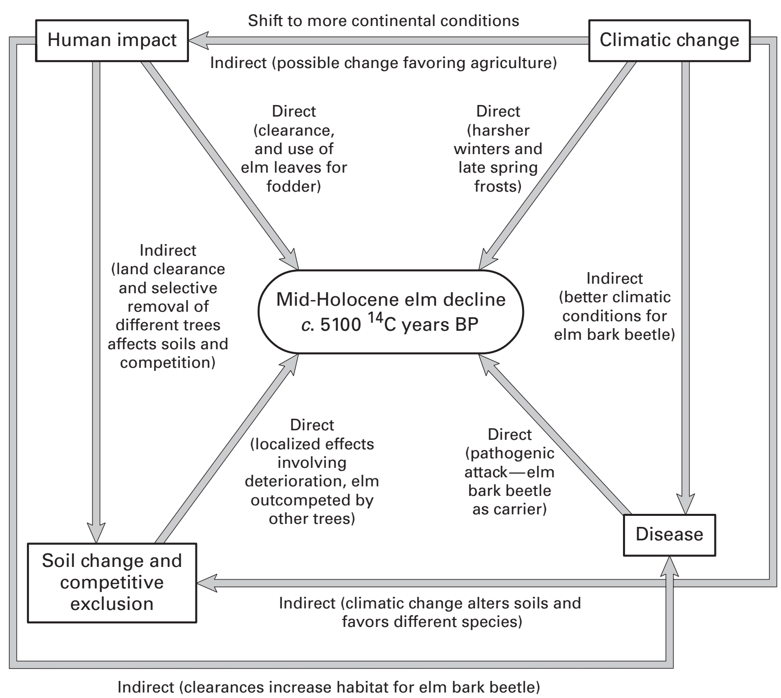

Figure 2.24 Relationships between factors influencing the mid-Holocene elm decline (after Parker et al., 2002, figure 9).

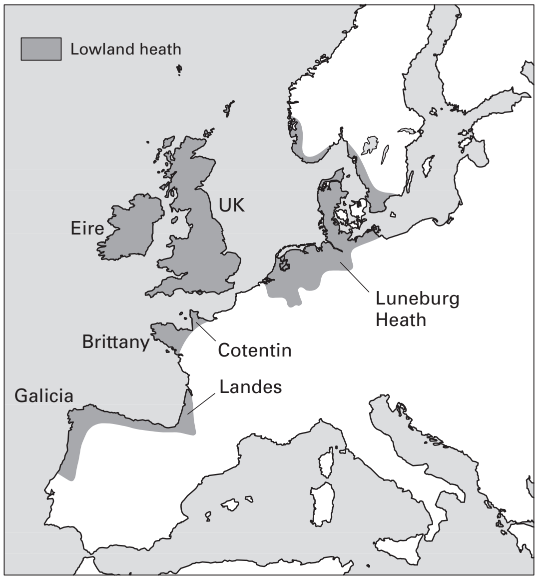

Figure 2.25 The lowland heath region of western Europe (modified after Gimingham and de Smidt, 1983, figure 2).

Lowland heaths

Heathland is characteristic of temperate, oceanic conditions on acidic substrates, and is composed of ericoid low shrubs, which form a closed canopy at heights that are usually less than 2 m. Trees and tall shrubs are absent or scattered. Some heathlands are natural: for example, communities at altitudes above the forest limit on mountains and those on exposed coasts. There are also well-documented examples of heath communities that appear naturally in the course of plant succession as, for example, where *Calluna vulgaris* (heather) colonizes *Ammophila arenaria* and *Carex arenaria* on coastal dunes.

However, at low and medium altitudes on the western fringes of Europe between Portugal and Scandinavia (Figure 2.25) extensive areas of heathland occur. The origin of these is strongly disputed (Gimingham and de Smidt, 1983). Some areas were once thought to have developed where there were appropriate edaphic conditions (e.g., well-drained loess or very sandy, poor soils), but pollen analysis showed that most heathlands occupy areas which were formerly tree-covered. This evidence alone, however, did not settle the question whether the change from forest to heath might have been caused by Holocene climatic change. However, the presence of human artifacts and buried charcoal, and the fact that the replacement of forest by heath has occurred at many different points in time between the Neolithic and the late nineteenth century, suggest that human actions established, and then maintained, most of the heathland areas. In particular, fire is an important management tool for heather in locations such as upland Britain, since the value of *Calluna* as a source of food for grazing animals increases if it is periodically burned.

The area covered by heathland in western Europe reached a peak around 1860, but since then there has been a very rapid decline. For example, by 1960 there had been a 60–70% reduction in Sweden and Denmark (Gimingham, 1981). Reductions in Britain averaged 40% between 1950 and 1984, and this was a continuation of a more long-term trend (Figure 2.26). The reasons for this fall are many, and include unsatisfactory burning practices, peat removal, drainage fertilization, replacement by improved grassland, conversion to forest, and sand and gravel abstraction. In England, the Dorset heathlands that were such a feature of Hardy's Wessex novels are now a fraction of their former extent.

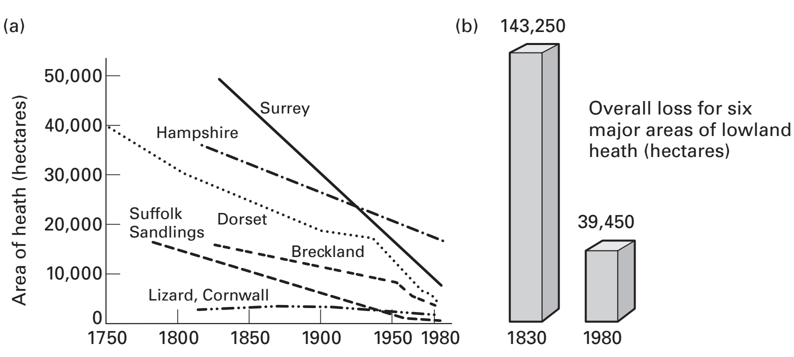

Figure 2.26 Losses of lowland heath in southern England (from Nature Conservancy Council, 1984).

Thus far we have considered the human impact on general assemblages of vegetation over broad zones. However, in turning to questions such as the range of individual plant species, the human role is no less significant.

Introduction, invasion, and explosion

People are important agents in the spread of plants and other organisms (Bates, 1956). Some plants are introduced deliberately by humans to new areas; these include crops, and ornamental and miscellaneous landscape modifiers (trees for reforestation, cover plants for erosion control, etc.). Indeed, some plants, such as bananas and breadfruit, have become completely dependent on people for reproduction and dispersal, and in some cases they have lost the capacity for producing viable seeds and depend on human-controlled vegetation propagation. Most cultigens are not able to survive without human attention, partly because of this low capacity for self-propagation, but also because they cannot usually compete with the better-adapted native vegetation.

However, some domesticated plants have, when left to their own devices, shown that they are capable of at least ephemeral colonization, and a small number have successfully naturalized themselves in areas other than their supposed region of origin (Gade, 1976). Examples of such plants include several umbelliferous annual garden crops (fennel, parsnip, and celery) which, though native to Mediterranean Europe, have colonized wastelands in California. The Irish potato, which is native to South America, grows unaided in the mountains of Lesotho. The peach (in New Zealand), the guava (in the Philippines), coffee (in Haiti), and the coconut palm (on Indian Ocean island strands) are perennials that have established themselves as wild-growing populations, although the last-named is probably within the hearth region of its probable domestication. In Paraguay, orange trees (originating in Southeast Asia and the East Indies) have demonstrated their ability to survive in direct competition with natural vegetation.

Plants that have been introduced deliberately because they have recognized virtues (Jarvis, 1979) can be usefully divided into an economic group (e.g., crops, timber trees, etc.) and an ornamental or amenity one. In the British Isles, Jarvis believes that the great bulk of deliberate introductions before the sixteenth century had some sort of economic merit, but that only a handful of the species introduced thereafter were brought in because of their utility. Instead, plants were introduced increasingly out of curiosity or for decorative value.

A major role in such deliberate introductions was played by European botanic gardens (Figure 2.27) and those in the colonial territories from the sixteenth century onwards. Many of the 'tropical' gardens (such as those in Calcutta, Mauritius, and Singapore) were often more like staging posts or introduction centers than botanic gardens in the modern sense (Heywood, 1989).

Figure 2.27 A monument in the botanic gardens in Mauritius, drawing attention to the value of introduced plants.

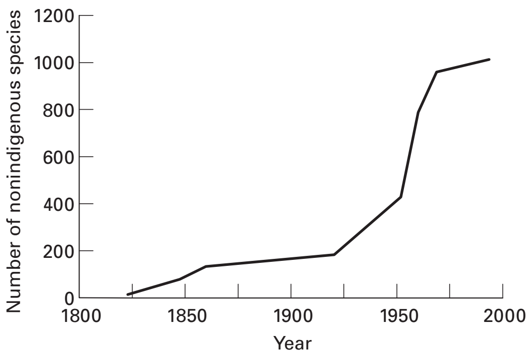

Figure 2.28 Number of nonindigenous plant species by date as reported in botanical treatments of the California flora (from Schwartz et al., 1996, figure 47.1).

Many plants, however, have been dispersed accidentally as a result of human activity: some by adhesion to moving objects, such as individuals themselves or their vehicles; some among crop seed; some among other plants (such as fodder or packing materials); some among minerals (such as ballast or road metal); and some by the carriage of seeds for purposes other than planting (as with drug plants). As illustrated in Figure 2.28, based on California, the establishment of alien species has proceeded rapidly over the past 150 years. In the Pampa of Argentina, Schmeider (1927b) estimates that the invasion of the country by European plants has taken place on such a large scale that at present only one-tenth of the plants growing wild in the Pampa are native.

The accidental dispersal of such plants and organisms can have serious ecological consequences (see Williamson, 1996, for a recent analysis). In Britain, for instance, many elm trees died in the 1970s because of the accidental introduction of the Dutch elm disease fungus which arrived on imported timber at certain ports, notably Avonmouth and the Thames Estuary ports (Sarre, 1978). There are also other examples of the dramatic impact of some introduced plant pathogens (von Broembsen, 1989). The American chestnut *Castanea dentata* was, following the introduction of the chestnut blight fungus *Cryphonectria parasitica* in ornamental nursery material from Asia late in the 1890s, almost eliminated throughout its natural range in less than 50 years. In western Australia the great jarrah forests have been invaded and decimated by a root fungus, *Phytophthora cinnamomi*. This was probably introduced on diseased nursery material from eastern Australia, and the spread of the disease within the forests was facilitated by road building, logging, and mining activities that involved movement of soil or gravel containing the fungus. More than three million hectares of forest have been affected.

In the USA, Kudzu (*Pueraria montana*) has been a particularly difficult invader. It was introduced into the USA from Japan in 1876 and was promoted as an ornamental and forage crop plant. From 1935 to the mid-1950s it was recommended as a good means of reducing soil erosion. Unfortunately this large vine, with massive tap roots and the ability to spread at *c*. 30 cm per day, now smothers large expanses of the southeastern USA. It is fiendishly difficult to eradicate because of the nature of its root system.

Ocean islands have often been particularly vulnerable to plant invasions. The simplicity of their ecosystems inevitably leads to diminished stability, and introduced species often find that the relative lack of competition enables them to broaden their ecological range more easily than on the continents. Moreover, because the natural species inhabiting remote islands have been selected primarily for their dispersal capacity, they have not necessarily been dominant or even highly successful in their original continental setting. Therefore, introduced species may prove more vigorous and effective (Holdgate and Wace, 1961). There may also be a lack of indigenous species to adapt to conditions such as bare ground caused by humans. Many successful invasive plants have escaped the natural enemies that hold them in check, freeing them to utilize their full competitive potential (Callaway and Aschehoug, 2000). Thus introduced weeds may catch on.

Table 2.6 illustrates clearly the extent to which the flora of selected islands now contain alien species, with the percentage varying between about one-quarter and two-thirds of the total number of species present.

There are a number of major threats that invasive plants pose to natural ecosystems (Levine et al., 2003). These have been discussed by Cronk and Fuller (1995):

1 replacement of diverse systems with single species stands of aliens, leading to a reduction in biodiversity, as for example where Australian acacias have invaded the fynbos heathlands of South Africa (Le Maitre et al., 2000);

Table 2.6 Alien plant species on oceanic islands. Source: from data in Moore (1983)

Island	*Number of native species*	*Number of alien species*	*Percent of alien species in flora*
New Zealand	1200	1700	58.6
Campbell Island	128	81	39.0
South Georgia	26	54	67.5
Kerguelen	29	33	53.2
Tristan da Cunha	70	97	58.6
Falklands	160	89	35.7
Tierra del Fuego	430	128	23.0

2 direct threats to native faunas by change of habitat;
3 alteration of soil chemistry (e.g., the African *Mesembryanthemum crystallinum* accumulates large quantities of salt, and in this way it salinizes invaded areas in Australia and may prevent the native vegetation from establishment);
4 alteration of geomorphic processes, especially rates of sedimentation and movement of mobile landforms (e.g., dunes and salt marshes);
5 plant extinction by competition;
6 alteration of fire regime (e.g., in Florida, USA the introduction of the Australasian *Melaleuca quinquenervia* has increased the frequency of fires because of its flammability, and has damaged the native vegetation which is less well adapted to fire);
7 alteration of hydrological conditions (e.g., reduction in groundwater levels caused by some species having high rates of transpiration).

The introduction of new animals can have an adverse effect on plant species. A clear demonstration of this comes from the atoll of Laysan in the Hawaii group. Rabbits and hares were introduced in 1903 in the hope of establishing a meat cannery. The number of native species of plants at this time was 25; by 1923 it had fallen to four. In that year all the rabbits and hares were systematically exterminated to prevent the island turning into a desert, but recovery has been slower than destruction. By 1930 there were nine species, and by 1961 sixteen species, on the island (Stoddart, 1968).

Another example is the extensive introduction of pigs to the islands of the Pacific. Long-established feral populations are known on many islands. Like rabbits they have caused considerable damage, not least because of their non-fastidious eating habits and their propensity for rooting into the soil. This is a theme that is reviewed by Nunn (1991).

It has often been proposed that the introduction of exotic terrestrial mammals has had a profound effect on the flora of New Zealand. Among the reasons that have been put forward for this belief are that the absence of native terrestrial mammalian herbivores permitted the evolution of a flora highly vulnerable to damage from browsing and grazing, and that the populations of wild animals (including deer and opossums) that were introduced in the nineteenth century grew explosively because of the lack of competitors and predators. It has, however, proved difficult to determine the magnitude of the effects that the introduced mammals had on the native forests (Veblen and Stewart, 1982).

Overall, invasive species may be one of the prime causes of loss of **biodiversity** because of their role in competition, predation, and **hybridization**, although this is a view that has been challenged by Theodoropoulos (2003). They also have enormous economic costs. On a global basis plant and animal invaders may cost as much as $1.4 trillion a year, representing 5% of the world economy (Pimentel, 2003).

There are many other examples of ecological explosions – bioinvasions – caused by humans creating new habitats. Some of the most striking are associated with the establishment of artificial lakes in place of rivers. Riverine species which cannot cope with the changed conditions tend to disappear, while others that can exploit the new sources of food, and reproduce themselves under the new conditions, multiply rapidly in the absence of competition (Lowe-McConnell, 1975). Vegetation on land flooded as the lake waters rise decomposes to provide a rich supply of nutrients, which allows explosive outgrowth of organisms as the new lake fills. In particular, floating plants that form dense mats of vegetation, which in turn support large populations of invertebrate animals, may cause fish deaths by deoxygenating the water, and can create a serious nuisance for turbines, navigators, and fishermen. On Lake Kariba in Central Africa there were dramatic growths in the communities of the South American water fern (*Salvinia molesta*), bladder-wort (*Utricularia*), and the African water lettuce (*Pistia stratiotes*); on the Nile behind the Jebel Aulia Dam there was a huge increase in the number of water hyacinths (*Eichhornia crassipes*); and in the Tennessee Valley lakes there was a massive outbreak of the Eurasian water-millfoil (*Myriophyllum*).

Roads have been of major importance in the spread of plants. As Frenkel (1970) has pointed out in a valuable survey of this aspect of anthropogenic biogeography:

> By providing a route for the bearers of plant propagules – man, animal, and vehicle – and by furnishing, along their margins, a highly specialized habitat for plant establishment, roads may facilitate the entry of plants into a new area. In this manner roads supply a cohesive directional component, cutting across physical barriers, linking suitable habitat to suitable habitat.

Roadsides tend to possess a distinctive flora in comparison with the natural vegetation of an area. As Frenkel (1970: 1) has again written:

> Roadsides are characterized by numerous ecological modification including: treading, soil compaction, confined drainage, increased runoff, removal of organic matter and sometimes additions of litter or waste material of frequently high nitrogen content (including urine and feces), mowing or crushing of tall vegetation but occasionally the addition of wood chips or straw, substrate maintained in an ecologically open condition by blading, intensified frost action, rill and sheetwash erosion, snow deposition (together with accumulated dirt, gravel, salt and cinder associated with winter maintenance), soil and rock additions related to slumping and rock-falls, and altered microclimatic conditions associated with pavement and right-of-way structures. Furthermore, road rights-of-way may be used for driving stock in which case unselective, hurried but often close grazing may constitute an additional modification. Where highway landscaping or stability of cuts and fills is a concern, exotic or native plants may be planted and nurtured.

The speed with which plants can invade roadsides is impressive. A study by Helliwell (1974) demonstrated that the M1 motorway in England, less than 12 years after its construction, had on its cuttings and embankments not only the 30 species that had been deliberately sown or planted, but also more than 350 species that had not been introduced there.

Railways have also played their role in plant dispersal. The classic example of this is provided by the Oxford ragwort, *Senecio squalidus*, a species native to Sicily and southern Italy. It spread from the Oxford Botanical Garden (where it had been established since at least 1690) and colonized the walls of Oxford. Much of its dispersal to the rest of Britain (Usher, 1973) was achieved by the Great Western Railway, in the vortices of whose trains and in whose cargoes of ballast and iron ore the plumed fruits were carried. The distribution of the plant was very much associated with railway lines, railway towns, and waste ground.

Indeed, by clearing forest, cultivating, depositing rubbish, and many other activities, humans have opened up a whole series of environments that are favorable to colonization by a particular group of plants. Such plants are generally thought of as weeds. In fact it has often been said that the history of weeds is the history of human society (although the converse might equally be true), and that such plants follow people like flies follow a ripe banana or a gourd of unpasteurized beer (see Harlan, 1975b).

One weed which has been causing especially severe problems in upland Britain in the 1980s is bracken (*Pteridium aquilinum*), although the problems it poses through rapid encroachment are of wider geographical significance (Pakeman et al., 1996). Indeed, Taylor (1985: 53) maintains that it ‘may justifiably be dubbed as the most successful international weed of the twentieth century’, for ‘it is found, and is mostly expanding, in all of the continents.’ This tolerant, aggressive opportunist follows characteristically in the wake of evacuated settlement, deforestation or reduced grazing pressure, and estimated encroachment rates in the upland parts of the UK average 1% per annum. This encroachment results from reduced use of bracken as a resource (e.g., for roofing) and from changes in grazing practices in marginal areas. As bracken is hostile to many other plants and animals, and generates toxins, including some carcinogens, this is a serious issue.

Air pollution and its effects on plants

Air pollutants exist in gaseous or particulate forms. The gaseous pollutants may be separated into primary and secondary forms. The primary pollutants, such as sulfur dioxide, most oxides of nitrogen, and carbon monoxide, are those directly emitted into the air from, for example, industrial sources. Secondary air pollutants, such as ozone and products of **photochemical** reactions, are formed as a consequence of subsequent chemical processes in the atmosphere, involving the primary pollutants and other agents such as sunlight. Particulate pollutants consist of very small solid- or liquid-suspended droplets (e.g., dust, smoke, and aerosolic salts), and contain a wide range of insoluble components (e.g., quartz) and of soluble components (e.g., various common cations, together with chloride, sulfate, and nitrate).

Some of the air pollutants that humans have released into the atmosphere have had detrimental impacts on plants (Yunus and Iqbal, 1996): sulfur dioxide, for example, is toxic to them. This was shown by Cohen and Rushton (in Barrass, 1974) who grew plants in containers of similar soil in different parts of Leeds, England, in 1913. They found a close relationship between the amount of sulfate in the air and in the plants, and in the yield obtained (Figure 2.29). In the more polluted areas of the city the leaves were blackened by soot and there was a smaller leaf area. The whole theme of urban vegetation has been studied in the North American context by Schmid (1975).

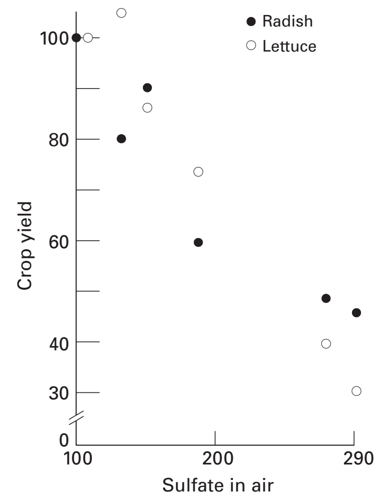

Figure 2.29 The effect of air quality on plant growth in Leeds, England, in 1913. Values for yield and for air sulfate in a low pollution area are taken as 100 and other values are scaled in proportion (data of Cohen and Rushton, in Barrass, 1974: 187).

Lichens are also sensitive to air pollution and have been found to be rare in central areas of cities such as Bonn, Helsinki, Stockholm, Paris, and London. There appears to be a zonation of lichen types around big cities as shown for northeast England (Figure 2.30a) and Belfast (Figure 2.30b). Moreover, when a healthy lichen is transplanted from the country to a polluted atmosphere, the algae component gradually deteriorates and then the whole plant dies (see Gilbert, 1970). Overall it has been calculated (Rose, 1970) that more than one-third of England and Wales, extending in a belt from the London area to Birmingham, broadening out to include the industrial Midlands and most of Lancashire and West Yorkshire, and reaching up to Tyneside, has lost nearly all its epiphytic lichen flora, largely because of sulfur dioxide pollution. Hawksworth (1990: 50), indeed, believes 'the evidence that sulphur dioxide is the major pollutant responsible

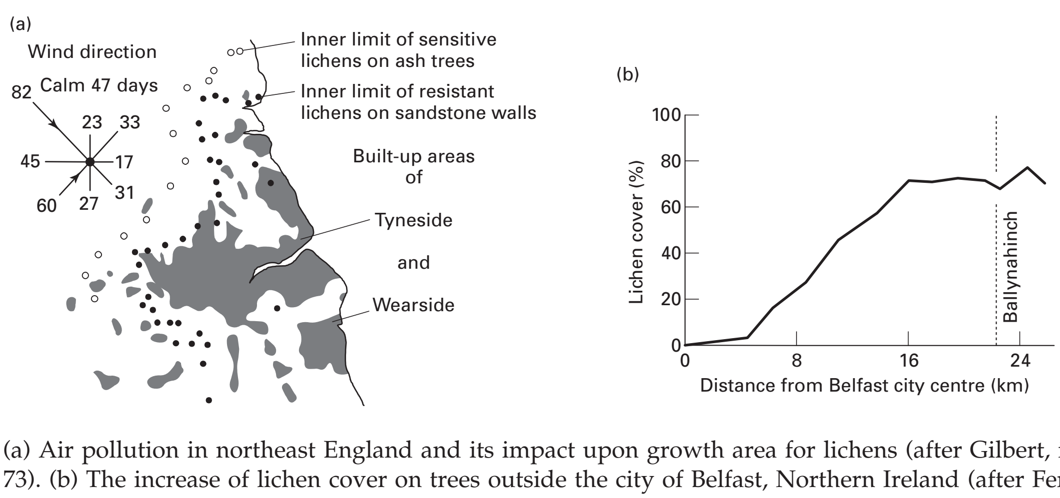

Figure 2.30 (a) Air pollution in northeast England and its impact upon growth area for lichens (after Gilbert, in Barrass, 1974, figure 73). (b) The increase of lichen cover on trees outside the city of Belfast, Northern Ireland (after Fenton, in Mellanby, 1967, figure 3).

Figure 2.31 The great smelter at Sudbury in Canada. Pollution derived from the fumes from such sources can have many deleterious impacts on vegetation in their neighborhood.

for impoverishment of lichen communities over wide areas of Europe is now overwhelming', but he also points out that sulfur dioxide is not necessarily always the cause of impoverishment. Other factors, such as fluoride or photochemical smog, acting independently or synergistically, can also be significant in particular locations.

Local concentrations of industrial fumes also kill vegetation (Freedman, 1995). In the case of the smelters of the Sudbury mining district of Canada (Figure 2.31), 2 million tonnes of noxious gases annually affect a 1900 km^2 area and white pine now exists only on about 7–8% of the productive land. Likewise in the lower Swansea Valley, Wales, the fumes from a century of coal burning resulted in almost complete destruction of the vegetation, with concomitant soil erosion. The area became a virtual desert. In Norway a number of the larger Norwegian aluminum smelters built immediately after the Second World War were sited in deep, narrow, steep-sided valleys at the heads of fjords. The relief has not proved conducive to the rapid dispersal of fumes, particularly of fluoride. In one valley a smelter with a production of 1,110,000 tonnes per year causes the death of pines (*Pinus sylveststris*) for over 13 km in each direction up and down the valley. To about 6 km from the source all pines are dead. Birch, however, seems to be able to withstand these conditions, and to grow vigorously right up to the factory fence (Gilbert, 1975).

Photochemical smog is also known to have adverse effects on plants both within cities and on their outskirts. In California, ponderosa pines (*Pinus ponderosa*) in the San Bernadino mountains as much as 129 km to the east of Los Angeles have been damaged extensively by smog and ozone (Diem, 2003). In summer months in Britain ozone concentrations produced by photochemical reactions have reached around 17 pphm (parts per hundred million) compared with a maximum of 4 pphm associated with clean air, while in Los Angeles ozone concentrations may reach 70 pphm (Marx, 1975). Fumigation experiments in the USA show that plant injury can occur at levels only marginally above the natural maximum and well within the summertime levels now known to be present in Britain and the USA. Ozone appears to reduce photosynthesis and to inhibit flowering and germination (Smith, 1974). It also seems to predispose conifers to bark-beetle infestation and to microbial pathogens.

Vegetation will also be adversely affected by excessive quantities of suspended particulate matter in the atmosphere. The particles, by covering leaves and plugging plant stomata, reduce both the absorption of carbon dioxide from the atmosphere and the intensity of sunlight reaching the interior of the leaf. Both tendencies may suppress the growth of some plants. This and other consequences of air pollution are well reviewed by Elsom (1992).

The adverse effects of pollution on plants are not restricted to air pollution: water and soil pollution can also be serious. Excessive amounts of heavy metals may prove toxic to them (Hughes et al., 1980) and, as a consequence, distinctive patterns of plant species may occur in areas contaminated with the waste from copper, lead, zinc, and nickel mines (Cole and Smith, 1984). Heavy metals in soils may also be toxic to microbes, and especially to fungi, which may in turn change the environment by reducing rates of leaf-litter decomposition (Smith, 1974). In many areas it has proved extremely difficult to undertake effective re-establishment of plant communities on mining spoil tips, although toxicity is only one of the problems (Kent, 1982).

Other types of industrial effluents may smother and poison some species. Salt marshes, mangrove swamps, and other kinds of wetlands are particularly sensitive to oil spills, for they tend to be anaerobic environments in which the plants must ventilate their root systems through pores or openings that are prone to coating and clogging (Lugo et al., 1981). The situation is especially serious if the system is not subjected to flushing by, for example, frequent tidal inundation. There are may case studies of the consequences of oil spills. For example, at the Fawley Oil Refinery on Southampton Water in England, Dicks (1977) has shown how the salt marsh vegetation has been transformed by the pollution created by films of oil, and how, over extensive areas, *Spartina anglica* has been killed off.

Forest decline

Forest decline, now often called *Waldsterben* or *Waldschäden* (the German words for 'forest death' and 'forest decline'), is an environmental issue that attained considerable prominence in the 1980s. The common symptoms of this phenomenon (modified from World Resources Institute, 1986, table 12.9) are:

1 *Growth-decreasing symptoms*:
- discoloration and loss of needles and leaves;
- loss of feeder-root biomass (especially in conifers);
- decreased annual increment (width of growth rings);
- premature aging of older needles in conifers;
- increased susceptibility to secondary root and foliar pathogens;
- death of herbaceous vegetation beneath affected trees;
- prodigious production of lichens on affected trees;
- death of affected trees.

2 *Abnormal growth symptoms*:
- active shedding of needles and leaves while still green, with no indication of disease;
- shedding of whole green shoots, especially in spruce;
- altered branching habit;
- altered morphology of leaves.

3 *Water-stress symptoms*:
- altered water balance;
- increased incidence of wet wood disease.

The decline appeared to be widespread in much of Europe and to be particularly severe in Poland and the Czech Republic (see Table 2.7). The process was also thought to be undermining the health of North America's high-elevation eastern coniferous forests (World Resources Institute, 1986, chapter 12). In Germany it was the white fir, *Abies alba*, which was afflicted initially, but since then the symptoms spread to at least ten other species in Europe, including Norway spruce (*Picea abies*), Scotch pine (*Pinus sylvestris*), European larch (*Larix decidua*), and seven broad-leaved species.

Many hypotheses have been put forward to explain this dieback (Wellburn, 1988; Federal Research Center for Forestry and Forest Products, 2000): poor forest management practices, aging of stands, climatic change, severe climatic events (such as the severe

Table 2.7 Results from forest damage surveys in Europe: percentage of trees with > 25% defoliation (all species). Source: data in *Acid News* (1995: 7)

	Mean of 1993–94		*Mean of 1993–94*
Austria	8	Latvia	33
Belarus	33	Lithuania	26
Belgium	16	Luxembourg	29
Bulgaria	26	Netherlands	22
Croatia	24	Norway	26
Czech Republic	56	Poland	52
Denmark	35	Portugal	7
Estonia	18	Romania	21
Finland	14	Slovak Republic	40
France	8	Slovenia	18
Germany	24	Spain	16
Greece	22	Switzerland	20
Hungary	21	UK	15
Italy	19		

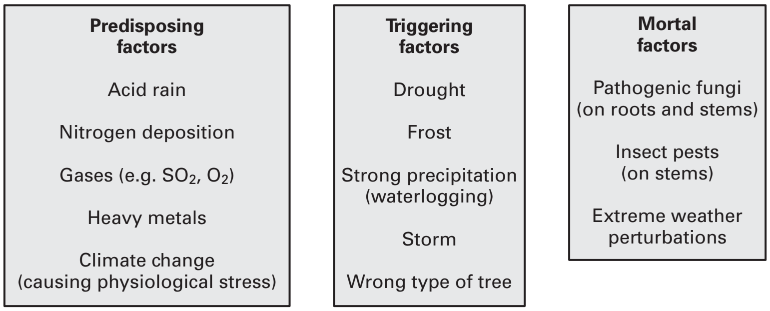

Figure 2.32 Some of the different stress variables used to explain forest decline (modified from Nihlgård, 1997, figure 24–1).

droughts in Britain during 1976), nutrient deficiency, viruses, fungal pathogens, and pest infestation. However, particular attention is being paid to the role of pollution, either by gaseous pollutants (sulfur dioxide, nitrous oxide, or ozone), acid deposition on leaves and needles, soil acidification and associated aluminum toxicity problems and excess leaching of nutrients (e.g., magnesium), overfertilization by deposited nitrogen, and trace metal or synthetic organic compound (e.g., pesticide, herbicide) accumulation as a result of atmospheric deposition.

The arguments for and against each of these possible factors have been expertly reviewed by Innes (1987), who believes that in all probability most cases of forest decline are the result of the cumulative effects of a number of stresses. He draws a distinction between predisposing, inciting, and contributing stresses (p. 25):

> Predisposing stresses are those that operate over long time scales, such as climatic change and changes in soil properties. They place the tree under permanent stress and may weaken its ability to resist other forms of stress. Inciting stresses are those such as drought, frost and short-term pollution episodes, that operate over short time scales. A fully healthy tree would probably have been able to cope with these, but the presence of predisposing stresses interferes with the tree's mechanisms of natural recovery. Contributing stresses appear in weakened plants and are frequently classed as secondary factors. They include attack by some insect pests and root fungi. It is probable that all three types of stress are involved in the decline of trees.

An alternative categorization of the stresses leading to forest decline has been proposed by Nihlgård (1997). He also has three classes of stress that have a temporal dimension: predisposing factors (related primarily to pollution and long-term climatic change); triggering factors (including droughts, frosts, and inappropriate management or choice of trees); and mortal factors such as pests, pathogenic fungi, and extreme weather events, which can lead to plant death (Figure 2.32).

As with many environmental problems, interpretation of forest decline is bedeviled by a paucity of long-term data and detailed surveys. Given that forest condition oscillates from year to year in response to variability in climatic stress (e.g., drought, frost, wind throw) it is dangerous to infer long-term trends from short-term data. There may also be differences in causation in different areas. Thus while widespread forest death in eastern Europe may result from high concentrations of sulfur dioxide combined with extreme winter stress, this is a much less likely explanation in Britain, where sulfur dioxide concentrations have shown a marked decrease in recent years. Indeed, in Britain Innes and Boswell (1990: 46) suggest that the direct effects of gaseous pollutants appear to be very limited.

It is also important to recognize that some stresses may be particularly significant for a particular tree species. Thus, in 1987 a survey of ash trees in Great Britain showed extensive dieback over large areas of the country. Almost one-fifth of all ash trees sampled showed evidence of this phenomenon. Hull and Gibbs (1991) indicated that there was an association between dieback and the way the land is managed around the tree, with particularly high levels of damage being evident in trees adjacent to arable land. Uncontrolled stubble burning, the effects of drifting herbicides, and the consequences of excessive nitrate fertilizer applications to adjacent fields were seen as possible mechanisms. However, the prime cause of dieback was seen to be root disturbance and soil compaction by large agricultural machinery. Ash has shallow roots and if these are damaged repeatedly the tree's uptake

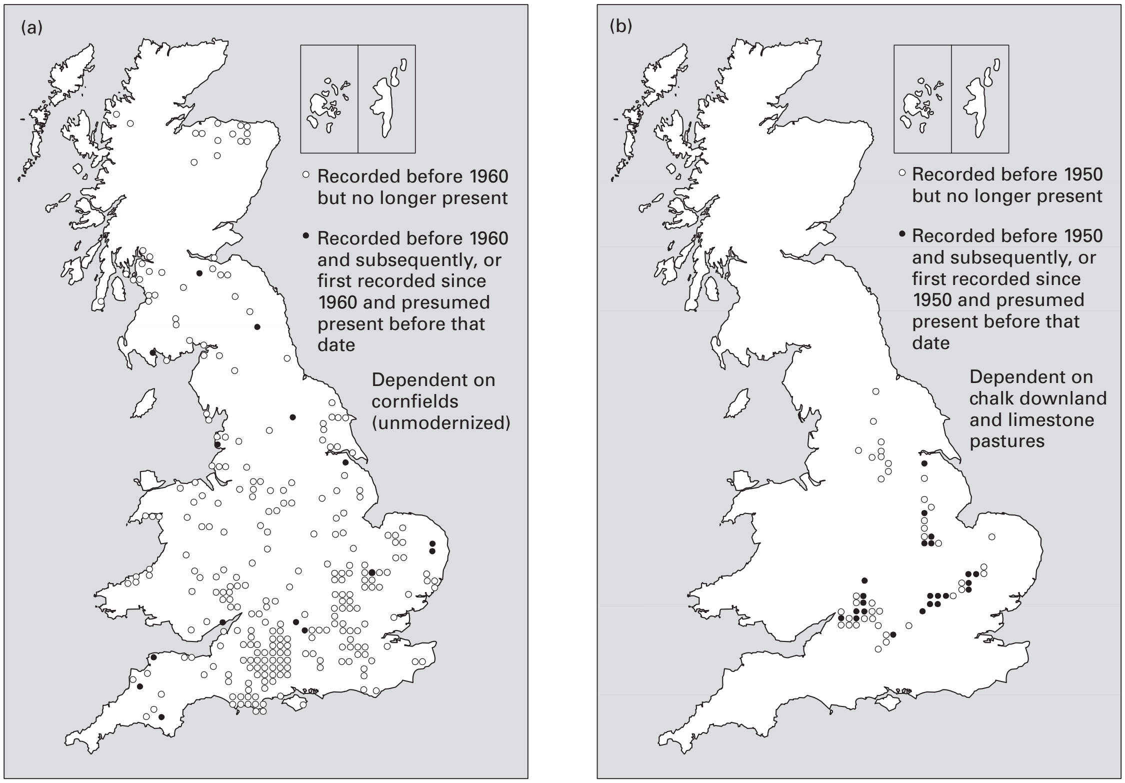

Figure 2.33 Reduction in the range of species related to habitat loss: (a) corncockle (*Agrostemma githago*); (b) pasque flower (*Pulsatilla vulgaris*) (after Nature Conservancy Council, 1977, figures 8 and 9).

of water and nutrients may be seriously reduced, while broken root surfaces would be prone to infection by pathogenic fungi.

Innes (1992: 51) also suggests that there has been some modification in views about the seriousness of the problem since the mid-1980s:

> The extent and magnitude of the forest decline is much less than initially believed. The use of crown density as an index of tree health has resulted in very inflated figures for forest 'damage' which cannot now be justified . . . If early surveys are discounted on the basis of inconsistent methodology . . . then there is very little evidence for a large-scale decline of tree health in Europe.
>
> . . . the term 'forest decline' is rather misleading in that there are relatively few cases where entire forest ecosystems are declining. Forest ecosystems are dynamic and may change through natural processes.

He also suggests that the decline of certain species has been associated with climatic stress for as long as records have been maintained.

Miscellaneous causes of plant decline

Some of the main causes of plant decline have already been referred to: deforestation, grazing, fire, and pollution. However, there are many records of species being affected by other forms of human interference. For example, casual flower-picking has resulted in local elimination of previously common species, and has been held responsible for decreases in species such as the primrose (*Primula vulgaris*) on a national scale in England. In addition, serious naturalists or plant collectors using their botanical knowledge to seek out rare, local, and unusual species can cause the eradication of rare plants in an area.

More significantly, agricultural 'improvements' mean that many types of habitat are disappearing or that the range of such habitats is diminishing. Plants associated with distinctive habitats suffer a comparable reduction in their range. This is certainly the case for two British plants (Figure 2.33a and b), the corncockle (*Agrostemma githago*) and the pasque flower (*Pulsatilla *vulgaris*). The former was characteristic of unmodified cereal fields, while the latter was characteristic of traditional chalk downland and limestone pastures. These calcareous grasslands were semi-natural swards formed by centuries of grazing by sheep and rabbits, and covered large areas underlain by Cretaceous chalk and Jurassic limestone until the Napoleonic Wars. Since then they have been subjected to increasing amounts of cultivation, reclamation and reseeding, and the loss of such grasslands between 1934 and 1972 has been estimated at around 75% (Nature Conservancy Council, 1984). Other plants have suffered a reduction in range because of drainage activities.

The introduction of pests, either deliberately or accidentally, can also lead to a decrease in the range and numbers of a particular species. Reference has already been made to the decline in the fortunes of the elm in Britain because of the unintentional establishment of Dutch Elm disease. There are, however, many cases where 'pests' have been introduced deliberately to check the explosive invasion of a particular plant. One of the most spectacular examples of this involves the history of the prickly pear (*Opuntia*) imported into Australia from the Americas (Figure 2.34). It was introduced some time before 1839 (Dodd, 1959), and spread dramatically. By 1900, it covered 4 million hectares, and by 1925 more than 24 million. Of the latter figure approximately one-half was occupied by dense growth (1200–2000 tonnes per hectare) and other more useful plants were excluded. To combat this menace one of *Opuntia's* natural enemies, a South American moth, *Cactoblastus*, was introduced to remarkable effect. By the year 1940 not less than 95% of the former 50 million acres (20 million hectares) of prickly pear in Queensland had been wiped out (Dodd, 1959: 575).

A further good example comes from Australia. By 1952 the aquatic fern, *Salvinia molesta*, which originated in southeastern Brazil, appeared in Queensland and spread explosively as a result of clonal growth, accompanied by fragmentation and dispersal. Significant pests and parasites appear to have been absent. Under optimal conditions *Salvinia* has a doubling time for biomass production of only 2.5 days. In June 1980 possible control agents from the *Salvinia's* native range in Brazil – the black, long-snouted weevil (*Cyrobagous* sp.) – were released on to Lake Moon Darra (which carried an infestation of 50,000 tonnes fresh weight of *Salvinia*, covering an area of 400 hectares). By August 1981 there was estimated to be less than 1 tonne of the weed left on the lake.

Figure 2.34 The prickly pear (*Opuntia*) is a plant that has been introduced from the Americas to Africa and Australia. It has often spread explosively. Recently it has been controlled by the introduction of moths and beetles, an example of biological control.

Finally, the growth of leisure activities is placing greater pressure on increasingly fragile communities, notably in tundra and high-altitude areas. These areas tend to recover slowly from disturbance, and both the trampling of human feet and the actions of vehicles can be severe (see e.g., Bayfield, 1979).

The change in genetic and species diversity

The application of modern science, technology, and industry to agriculture has led to some spectacular progress in recent decades through such developments as the use of fertilizers and the selective breeding of plants and animals. The latter has caused some concern, for in the process of evolution domesticated plants have become strikingly different from their wild progenitors. Plant species that have been cultivated for a very long time and are widely distributed demonstrate this particularly clearly. Crop evolution through the millennia has been shaped by complex interactions reflecting the pressures of both artificial and natural selection. Alternate isolation of stocks followed by migration and seed exchanges brought distinctive stocks into new environments and permitted new hybridizations and the recombination of characteristics. Great genetic diversity resulted.

There are fears, however, that since the Second World War the situation has begun to change (Harlan, 1975a). Modern plant-breeding programs have been established in many parts of the developing world in the midst of genetically rich centers of diversity. Some of these programs, associated with the so-called **Green Revolution**, have been successful, and new, uniform high-yielding varieties have begun to replace the wide range of old, local strains that have evolved over the millennia. This may lead to a serious decline in the genetic resources that could potentially serve as reservoirs of variability. Ehrlich et al. (1977: 344) have warned:

> Aside from nuclear war, there is probably no more serious environmental threat than the continued decay of the genetic variability of crops. Once the process has passed a certain point, humanity will have permanently lost the coevolutionary race with crop pests and diseases and will no longer be able to adapt crops to climatic change.

New, high-yielding crop varieties need continuous development if they are to avoid the effects of crop pests, and Ehrlich and Ehrlich (1982: 65) have summarized the situation thus:

> The life of a new cultivated wheat variety in the American Northwest is about five years. The rusts (fungi) adapt to the strain, and a new resistant one must be developed. That development is done through artificial selection: the plant breeder carefully combines genetic types that show promise of giving resistance.

The impact of human activities on species diversity, while clearly negative on a global scale (as evidenced by extinction rates) is not so cut-and-dried on the local scale. Under certain conditions chronic stress caused by humans can lead to extremely high numbers of coexisting species within small areas. By contrast, site enrichment or fertilization can result in a decline of species density (Peet et al., 1983).

Certain low productivity grasslands which have been grazed for long periods have high species densities in Japan, the UK and The Netherlands. The same applies to Mediterranean scrub vegetation in Israel, and to savanna in Sri Lanka and North Carolina (USA). Studies have confirmed that species densities may increase in areas subject to chronic mowing, burning, domestic grazing, rabbit grazing, or trampling. It is likely that in such ecosystems humans encourage a high diversity of plant growth by acting as a 'keystone predator', a species which prevents competitive exclusion by a few dominant species.

Grassland enrichment experiments employing fertilizers have suggested that in many cases high growth rates result in the competitive exclusion of many plants. Thus the tremendous increase in fertilizer use in agriculture has led to, with what in retrospect is the predictable result, a widespread decrease in the species diversity of grasslands.

One other area in which major developments will take place in the coming years, which will have implications for plant life, is the field of **genetic engineering**. This involves the manipulation of **DNA**, the basic chromosomal unit that exists in all cells and contains genetic information that is passed on to subsequent generations. Recombinant DNA technology (also known as *in vitro* genetic manipulation and gene cloning) enables the insertion of foreign DNA (containing the genetic information necessary to confer a particular target characteristic) into a vector. The advent of modern techniques of genetic modification (GM) enables the removal of individual genes from one species and their insertion into another, without the need for sexual compatibility. Once the new gene has been inserted, offspring that will contain copies of that new gene can be produced in the traditional manner. For example, a bacterial gene can be inserted into maize to give it resistance to certain insect pests. Indeed, genetic engineering is vastly different from conventional plant breeding methods because it allows scientists to insert a gene or genes from virtually any organism into any other organism. Fears have been expressed that this technology could produce pathogens that might interact detrimentally with naturally occurring species (see, e.g., Beringer, 2000; Letourneau and Burrows, 2001; Hails, 2002).

It has also been feared that GM could make a plant more invasive, leading to a new threat from superweeds. Equally, some genetically modified crops might be highly effective at controlling or eliminating weeds, thereby disrupting essential food sources for a wide range of organisms. Conversely, genetically modified herbicide-tolerant crops would bring greater economy and feasibility in weed control, so that fewer environmentally damaging chemical herbicides would be used than in conventional farming (Squire et al., 2003). Other beneficial traits from GM technology include resistance to insects or pathogens, drought resistance and salt tolerance, the ability to fix nitrogen, and so forth. There is an urgent need to evaluate the costs and benefits of this new technology, for already GM crops are being widely grown. The number of hectares planted to them in the USA increased from 1.4 million in 1996 to around 40 million in 2003. An estimated 38 trillion GM plants have now been grown in American soil!

Points for review

How does the use of fire have an impact upon the environment?

In what ways does grazing by domestic stock modify vegetation?

Where is present day deforestation taking place and why?

What controversies surround desertification?

What role have humans played in the development of savannas and mid-latitude grasslands?

What are ecological explosions?

How does pollution affect plants?

Guide to reading

Crutzen, P. J. and Goldammer, J. G., 1993, *Fire in the environment*. Chichester: Wiley. A particularly useful study of fire's importance.

Grainger, A., 1990, *The threatening desert: controlling desertification*. London: Earthscan. A very readable and wide-ranging review of desertification.

Grainger, A., 1992, *Controlling tropical deforestation*. London: Earthscan. An up-to-date introduction with a global perspective.

Grove, A. T. and Rackham, O., 2002, *The nature of Mediterranean Europe: an ecological history*. New Haven and London: Yale University Press. A well illustrated account of natural and human changes in the Mediterranean lands.

Holzner, W., Werger, M. J. A., Werger, I., and Ikusima, I. (eds), 1983, *Man's impact on vegetation*. Hague: Junk. A wide-ranging edited work with examples from many parts of the world.

Manning, R., 1995, *Grassland: the history, biology, politics and promise of the American prairie*. New York: Viking Books. A 'popular' discussion of the great grasslands of North America.

Meyer, W. B. and Turner, B. L. (eds), 1994, *Changes in land use and land cover: a global perspective*. Cambridge: Cambridge University Press. An excellent edited survey of human transformation of the biosphere.

Pimentel, D. (ed.), 2003, *Biological invasions*. Washington, DC: CRC Press. An edited, international account of invasive species.

Simmons, I. G., 1979, *Biogeography: natural and cultural*. London: Arnold. A useful summary of natural and human impacts on ecosystems.

Stewart, C. N., 2004, *Genetically modified planet. Environmental impacts of genetically engineered plants*. Oxford: Oxford University Press. A readable review of the costs and benefits of GM plants.

Williams, M., 2003, *Deforesting the earth. from prehistory to global crisis*. Chicago: University of Chicago Press. A magisterial overview of the global history of deforestation.