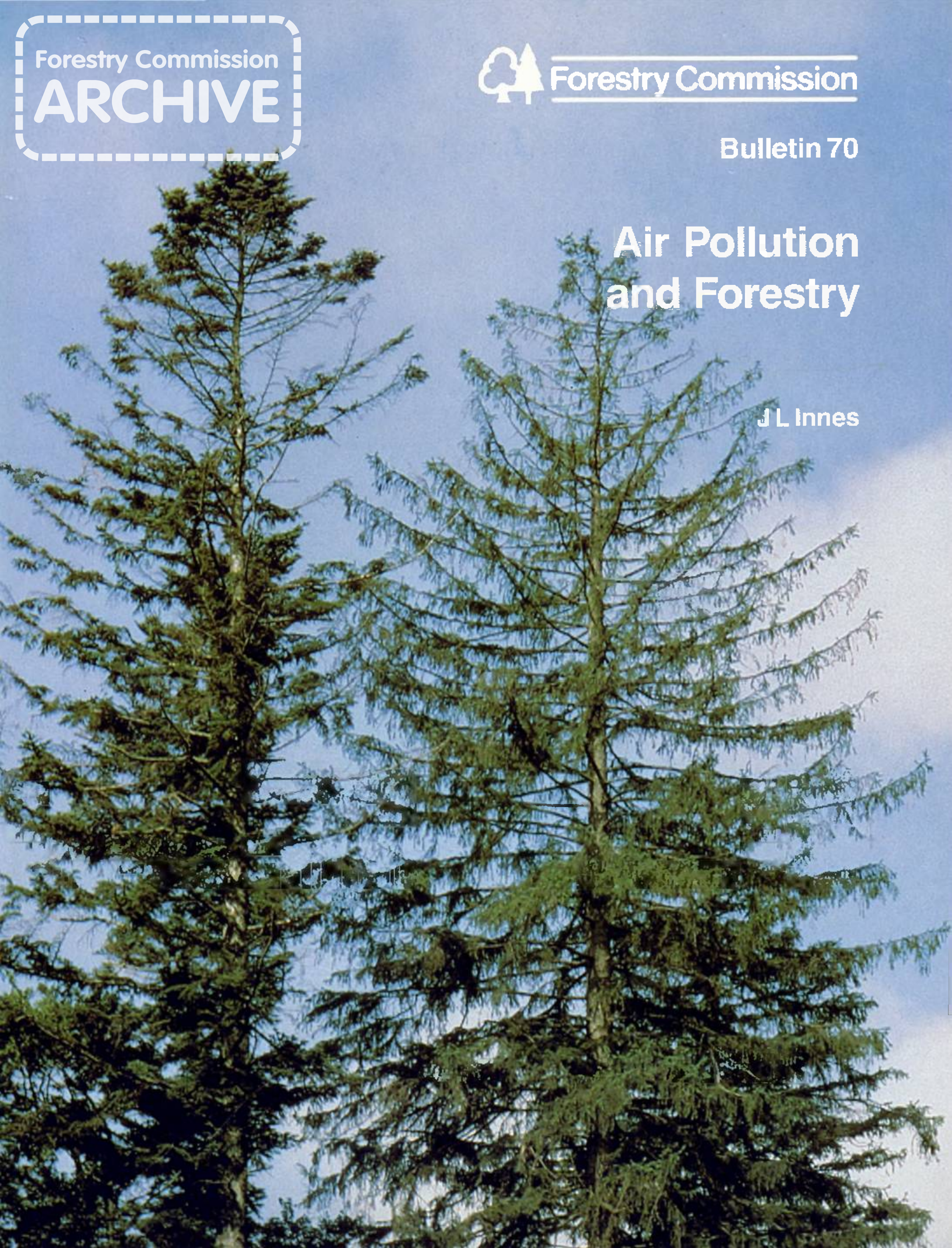
Forestry Commission

Bulletin 70

Air Pollution and Forestry

J L Innes



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Air Pollution and Forestry

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FRONT COVER: Air pollution is alleged to be the cause of the decline of forests over much of Europe. One of the most characteristic features of the decline is the loss of foliage within the crown, well-illustrated by these trees (*left*: Silver fir, *right*: Norway spruce) in the Bavarian National Park, West Germany (August 1986). (*J.L. Innes*)

Contents

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Introduction

The idea that air pollution ('acid rain') might destroy the trees and forests of Europe has been causing much concern. Although damage to trees by air pollution is long known, it was until recently held to be a local problem only. Such damage is still important in some places. However, more and more people have come to believe that long-range air pollution is responsible, in part at least, for the widespread decline of trees seen in many countries since the late 1970s. There is no doubt that the damage in countries such as West Germany is serious and probably unprecedented but, as yet, the evidence linking this decline with air pollution remains controversial.

As in many scientific fields where advances are rapid, there is much speculation about the causes of the decline. Theories abound, some more plausible than others. The aim of this Bulletin, which replaces an earlier paper[15], is to summarise the current information available on the interactions between air pollution and forests. While it is primarily concerned with Great Britain, air pollution is an international problem, so information from other countries has been included. Pollution from point sources, such as aluminium smelters and brickworks, has not been included, and this paper deals mainly with long-range pollution and its possible regional-scale effects.

In the following pages, a technical account of the formation of atmospheric pollutants is given. This is followed by a more general discussion of acidification and its effects on the environment. The symptoms shown by trees involved in the forest decline seen on the continent are described together with data from various national surveys; a major part of the report is concerned with the possible causes of this decline and its economic implications. Finally, research being undertaken by the Forestry Commission into the possible effects of air pollution on trees is described.

Long-distance Pollutants and their Sources

Long-range air pollutants can be divided into two groups: primary and secondary. Primary pollutants, such as sulphur dioxide (SO_2), nitric oxide (NO), hydrogen chloride (HCl), carbon monoxide (CO) and hydrocarbons (largely solvents and unburnt fuels), are emitted directly into the lower atmosphere. Secondary pollutants, such as 'acid rain' and ozone (O_3), are formed when primary pollutants and other atmospheric substances react in the presence of sunlight or water or both. Much information is available on the sources of primary pollutants; the most recent data for the United Kingdom are given in Table 1. The sources of the main pollutants are gradually changing. For example, power stations are responsible for a steadily increasing proportion of the total emissions of SO_2 (Figure 1).

Figure 1. Relative importance of power stations and industrial sources of SO_2 over the period 1974-1984. (Source: *Digest of Environmental Protection and Water Statistics,* No. 8, 1985).

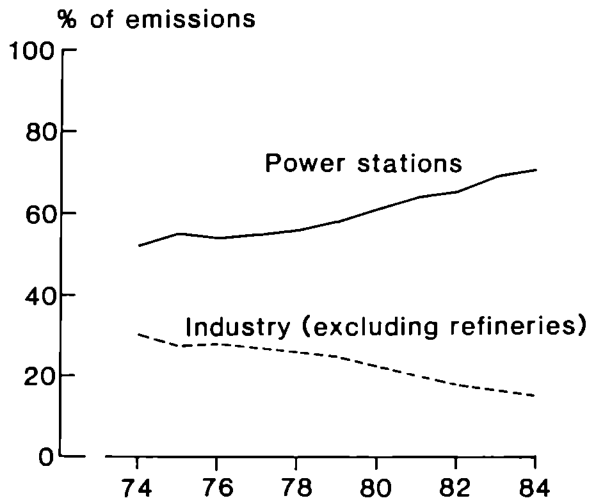
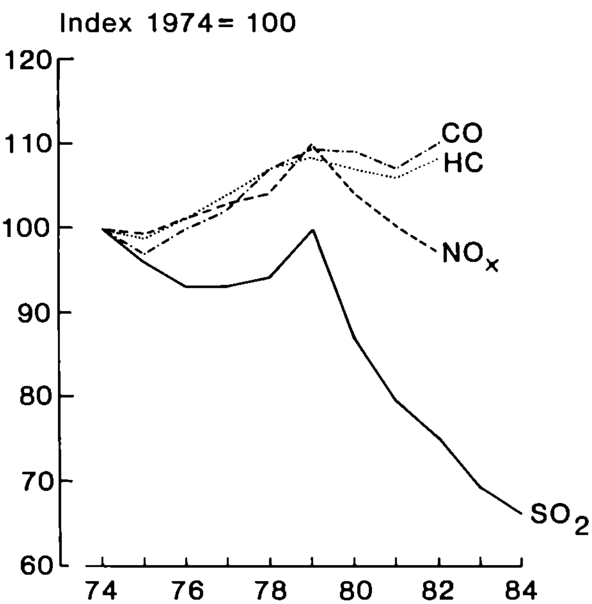


Table 1. Sources of pollutants in the United Kingdom (from *Digest of Environmental Protection and Water Statistics 1986*, HMSO). Data for 1983 and 1984 only are presented as the figures for nitrogen oxides, carbon monoxide and hydrocarbons are calculated using new emission factors which are not directly compatible with data from previous years. All figures in million tonnes per annum.

	SO_2		NO_x		CO		HC	
	1983	1984	1983	1984	1983	1984	1983	1984
Domestic	0.20	0.16	0.05	0.05	0.46	0.37	0.07	0.05
Commercial	0.14	0.14	0.04	0.04	0.01	0.01	*	*
Power stations	2.53	2.50	0.76	0.62	0.05	0.05	0.01	0.01
Refineries	0.16	0.15	0.04	0.04	*	*	*	*
Agriculture	0.01	0.01	*	*	*	*	n/a	n/a
Other industry	0.61	0.52	0.19	0.17	0.07	0.07	*	*
Rail transport	0.01	0.01	0.04	0.04	0.01	0.01	0.01	0.01
Road transport	0.04	0.04	0.69	0.72	4.44	4.46	0.53	0.54
Incineration/burning	n/a	n/a	0.01	0.01	0.22	0.22	0.04	0.04
Gas leakage	n/a	n/a	n/a	n/a	n/a	n/a	0.37	0.38
Industrial processes and solvent evaporation	n/a	n/a	n/a	n/a	n/a	n/a	0.60	0.60
Total	3.69	3.54	1.82	1.69	5.27	5.19	1.63	1.64

* less than 10 000 tonnes per annum.
n/a = not applicable.

Figure 2. Trends in emissions of CO, SO_2, NO_x and hydrocarbons in Britain during the period 1974-1984. Data for CO, NO_x and hydrocarbons for 1983 and 1984 are not comparable to earlier data due to changes in the method of calculating emissions and are therefore not given.
(Source: *Digest of Environmental Protection and Water Statistics*, No. 8, 1985).

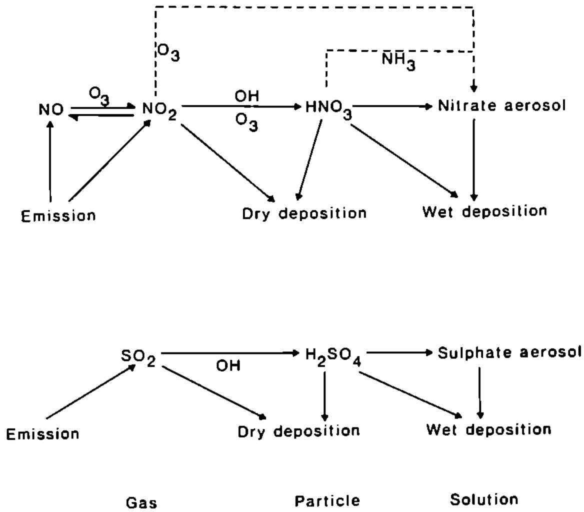


Recent trends in the emissions of the four main pollutants are given in Figure 2.

Many 'pollutants' also have natural sources such as volcanoes and sea spray. However, while about 40 per cent of the global sulphur emissions appear to be man-made[41], in the northern hemisphere the figure is thought to be nearer 90 per cent[138]. Much of this sulphur remains within the northern hemisphere, where it is eventually deposited. Global comparisons between man-made and natural emissions of gases such as SO_2 are misleading as both sources are unevenly distributed around the world. The relative proportions of man-made and natural sulphur in the atmosphere are dependent on the location being investigated. For instance more than 60 per cent of the sulphate being deposited in western Scotland is of marine (i.e. predominantly natural) origin, whereas the corresponding figure in the east of Scotland is only 10 per cent[62].

The ways in which some secondary pollutants, such as sulphate (SO_4), are formed is still uncertain. A highly simplified diagram of the main pathways for SO_2 and NO is given in Figure 3. Many of the reactions involve photochemical oxidants. These are chemicals that react with other materials in the atmosphere in the presence of sunlight. The most important are ozone, the hydroxyl radical (OH) and hydrogen peroxide (H_2O_2). Some O_3 is natural and some man-made. It occurs naturally at high altitudes (12-50 km) in the

Figure 3. Atmospheric pathways of SO_2 and NO. Night-time pathways are indicated by hatched (– – –) line. (Partly based on Derwent, R.G. and Nodop, K. (1986). Long-range transport and deposition of acidic nitrogen species in north-west Europe. *Nature* **324,** 356-358).



stratosphere, and it may enter the lower atmosphere during periods of atmospheric turbulence. In some special conditions, for example in association with cold fronts[151], stratospheric O_3 may reach the lowermost layer of the atmosphere resulting in quite high concentrations (more than 100 parts per billion [ppb])[46]. Such incursions appear to be rare. Although some O_3 may be formed in the lower atmosphere, most is believed to originate as a result of man's activities[45]. The usual mode of formation of O_3 at low altitudes is the interaction of nitrogen dioxide (NO_2) with sunlight and although it readily combines with NO to reform NO_2, in the presence of degraded hydrocarbons or other pollutants the balance between NO_2, NO and O_3 is broken down, resulting in a net build-up of O_3.

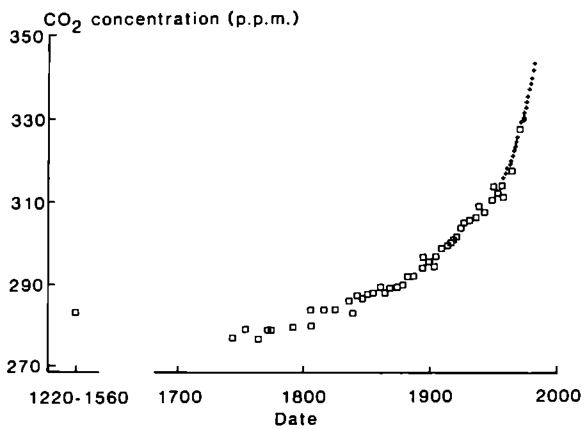
There are many other secondary pollutants in the atmosphere. Some of these, such as peroxyacetylnitrate (PAN) are known to affect plants adversely[174]. The concentrations that many of these pollutants reach, and their effects on plants, have yet to be determined.

A primary pollutant that has not been mentioned so far is ammonia (NH_3). This has been implicated in the dieback of trees in the Netherlands[25] and in Sweden[12,129]. Ammonia released into the air is rapidly depleted by deposition downwind and conversion into ammonium ions (NH_4^+). Although ammonia is an alkaline gas, its transformation into nitrate releases hydrogen ions, resulting in an increase in the acidity of the soil. Ammonia is considered to be the most important alkaline gas commonly found in the atmosphere[197] and it may act to neutralize some of the more acidic gases, increasing the rate of oxidation of SO_2 by an order of magnitude or more (through the production of ammonium sulphate). This may be significant: in Switzerland it is estimated that 50-60 per cent of the total deposited acidity is neutralized by ammonium[68]. Total ammonia emissions in the United Kingdom are thought to be about 400 000 tons per year, about 80 per cent of which is derived from agricultural sources[25]. The majority of agricultural ammonia production occurs as a result of spreading slurry on fields. Volatilization of ammonia is thought to result in the losses of several hundred kilograms of nitrogen per hectare per application[20]. The pattern of emissions therefore reflects the distribution of livestock; emissions in Britain are considered to be greatest in the south-west of the country.

Carbon dioxide

Carbon dioxide (CO_2) is a pollutant that does not readily fit into the primary and secondary classification. Concentrations of CO_2 are known to have risen over the last 200 years primarily as a result of burning fossil fuels, but also as a result of the release of CO_2 following deforestation, cultivation and soil destruction[94]. The trend in the atmospheric concentration of CO_2 over the past 200 years, derived from measurements from ice cores and direct measurements of atmospheric concentrations[64], is shown in Figure 4. Although the effects of an increase of CO_2 are unclear, most people believe that it will result in increased global

Figure 4. Rise in atmospheric CO_2 concentrations over the last 200 years (from Friedli, H., Lötscher, H., Oeschger, H., Siegenthaler, U. and Stauffer, B. (1986). Ice core record of the $^{13}C/^{12}C$ ratio of atmospheric CO_2 in the past two centuries. *Nature* **324,** 237-238).



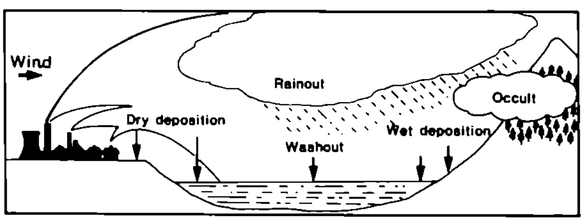
temperatures, and there is now evidence that these have indeed increased by about 0.5°C over the last 120 years[101]. Plants are also known to respond directly to increased CO_2 concentrations with increased growth, and the effect of increased CO_2 levels is likely to be the better growth of trees. The long-term impacts of such changes on factors such as nutrient cycling remain unclear.

Deposition of Primary and Secondary Pollutants

Deposition of pollutants occurs in two forms: dry and wet. The proportions of these depend on the distance from the source and the meteorological conditions at a site (Figure 5). Dry deposition, which is the fall-out of gaseous and particulate pollutants, mostly occurs close to the source of the pollution, although with secondary pollutants there may be a peak in deposition some distance from the source because of the time taken for the pollutant to form. Wet deposition is the deposition of pollutants held in solution and also includes the removal of particulate and gaseous materials by raindrops as they pass through the air. Wet deposition tends to be relatively more important further away from the source of the pollutants. Water droplets associated with cloud and fog can be deposited on vegetation and other surfaces without it actually raining. This is called occult deposition and it may represent a significant means of transferring pollutants from the atmosphere to vegetation in areas with a high frequency of cloud or fog. Wet deposition can even occur in bright sunshine, manifesting itself in the form of heat haze.

Deposition rates are normally expressed in terms of grams of nitrogen or sulphur per square metre per year. Alternatively, they may be expressed as kilograms per hectare per year. These do not indicate the deposited acidity, which is best given in terms of grams of hydrogen ions per square metre per year ($g\,H^+\,m^{-2}\,yr^{-1}$). In Britain, as in most countries, the small number of rural monitoring sites has meant that

Figure 5. Deposition processes of atmospheric acidity.



rates have had to be estimated, principally by the use of computer modelling, although in 1985 a network of precipitation composition monitoring sites was established. The results for 1986 are not yet available but, when published, must be viewed with caution. This is because the collectors do not obtain a representative sample of the rainfall. In particular, smaller raindrops tend to be excluded, and this may mean that the total amounts of deposited acidity are being underestimated by between 5 and 20 per cent[155,156]. The extent to which this affects the results will depend on the nature of the wind at the site: in windier places (such as at higher altitudes), there will be a greater underestimation of the deposition rates. In addition, there is now evidence that the concentration of pollutants in raindrops is at least partly dependent on the size of the raindrop[2], producing a further source of error as a result of the bias towards the collection of larger particles. Consequently, care must be taken in interpreting both modelled predictions of deposition and measured patterns.

'Acid Rain'

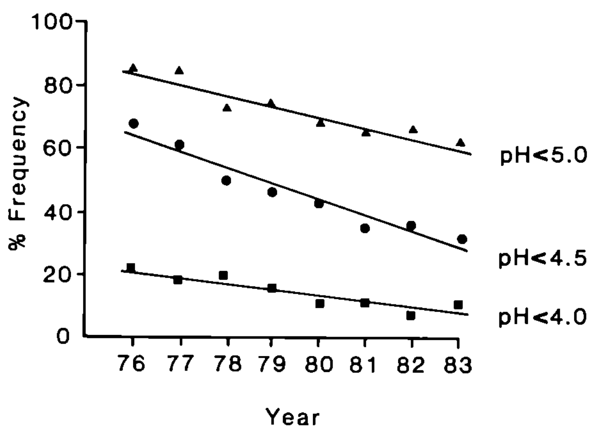
The acidity of rainwater is determined by the balance between cations (positively charged ions) and anions (negatively charged ions) in the rain. In Europe, the main cations are H^+ and NH_4^+ and, to a lesser extent, Ca^{2+}, Na^+, K^+ and Mg^{2+} and the main anions are SO_4^{2-}, NO_3^- and Cl^- [10]. It has been argued from a number of studies that the natural acidity of unpolluted rainwater is pH 5.6. This is now considered to be wrong[34] as measurements of the pH of rainfall from remote, supposedly unpolluted sites, indicate that the acidity can vary from pH 4.5 to above pH 5.6. The reason for this is the natural occurrence of some acidifying compounds. For example, in Cumbria, rainfall derived from the North Atlantic has a pH of about 5.0 (whereas rainfall derived from air masses that have passed over industrial areas has a pH of about 4.0)[61]. Background levels are thus extremely variable, depending on factors such as the distance from the sea and the proximity to other natural sources of SO_2.

Acidic precipitation is not a new phenomenon. There are reports of black snow in Scotland dating back to 1862[22]. In the Pennines, the occurrence of acidic precipitation was recognised throughout the 19th and first half of the 20th centuries. The precipitation, in combination with the very high concentrations of SO_2 in the air, resulted in substantial changes to the flora with many species of mosses, lichens and higher plants being lost following the Industrial Revolution. In the early 1980s, the annual mean pH of rainfall in the Pennines has been about pH 4.1[136].

Recorded rainfall acidities may be 10 times higher in polluted than in less-polluted areas of the same country. In Great Britain, rainfall with a pH of 3.0 to 4.0 appears to be relatively common. It is doubtful whether pH is a suitable index of rainfall acidity as it only measures the free protons (i.e. the hydrogen ions), not the total amount of acids. Nor does it measure all the constituents that can form acids in the soil. For example, NH_4^+ ions are not measured by pH. Attempts to evaluate trends in the acidity of rainfall over Europe have shown that there has been no significant increase over the last 20 years[104]. However, both SO_4^{2-} and NO_3^- concentrations in rainfall over Europe have increased over the same period[104].

Attempts to relate the reductions of emissions to trends in the acidity of precipitation have met with little success. This may be the result of a number of factors, the most important probably being the lack of data that can be used to establish such a relationship. Reductions in emissions must eventually result in a reduction of the total amount of deposited material, but the decrease may be far from clear on a regional scale. Some evidence is available for Britain, where reductions in the emissions of sulphur over the last 15 years have been accompanied by a reduction in acidity over the last 6 years[60], although the relationship is far from clear. In addition, there is evidence for a decline in the frequency of acidic (pH <5.0) rainfalls at Pitlochry in central Scotland over the past 10 years (Figure 6)[79]. In the United States, a longer series of data exists, and a positive correlation has been found between the emissions of sulphur dioxide and the concentrations of sulphate in streams in eastern States for the period 1967 to 1980[116,117]. Correlations between the emissions of sulphur dioxide and the deposition of sulphate have also been found in the western United States[50].

Figure 6. Variations in the % frequency of daily pH values of precipitation at Pitlochry. (Source: Harriman, R. and Wells, D.E. (1985). Causes and effects of surface water acidification in Scotland. *Journal of Water Pollution Control* **84**, 215-224).



Detailed records are available for precipitation chemistry at Hubbard's Brook in the eastern United States for the period 1964 to 1983[82]. While decreases have occurred in sulphate deposition, there has not been a concomitant decrease in acidity as a result of increases in nitrate concentrations. This latter study illustrates both the importance of looking at the total ion budget and a possible cause of the lack of linearity between SO_2 emissions and precipitation acidity.

The acidity of fog and cloud water may be up to 20 times higher than that of rainwater, with pH values of cloud water of 2.8 to 3.1 having been recorded[193]. In addition, the concentrations of sulphate and nitrate may be up to 70 times higher[67]. Unfortunately, the only similar data that appear to have been published for the UK at the present time are based on a single day's collection[48], and therefore cannot be extrapolated to longer time intervals. As discussed below, the acidity of fog and mist may be sufficiently high to cause direct injury to trees.

Acidification of Forest Soils

The acidity of a soil is highly complex and cannot be assessed by measurement of pH alone. In terms of acidic deposition, the buffer capacity or acid neutralizing capacity of a soil is a more useful measure than the pH[21]. Soil acidification is best defined as a progressive decrease in its buffer capacity and is caused by the removal of cations from the soil and, to a lesser extent, by the addition of anions. The buffer capacity is measured in terms of the number of mols or milliequivalents (meq) of hydrogen or hydroxyl ions that must be added to raise or lower the pH of one kilogram of soil by 1 pH unit[53]. As such, it has a much clearer relationship with the addition or depletion of protons in the soil than does pH.

The problems in defining soil acidification mean that there are few good records of long-term trends in soil acidity. Those that exist refer to pH and, in view of the reservations mentioned above, are of limited value. One of the best series is for 100 years at Rothamsted in Hertfordshire, England[100]. The soil acidity has increased through time, although the reasons for this are unclear.

In south-west Sweden, trends in the pH of soil have been measured over the period from 1927 to 1984[77].

Figure 7. Arithmetic mean pH in 1927 and 1982/1983 for different soil layers in spruce and beech stands at Tönnersjöheden Experimental Forest, Sweden. (Source: Hallbäcken, L. and Tamm, C.O. (1986). Changes in soil acidity from 1927 to 1982-1984 in a forest area of south-west Sweden. *Scandinavian Journal of Forest Research* **1**, 219-232).

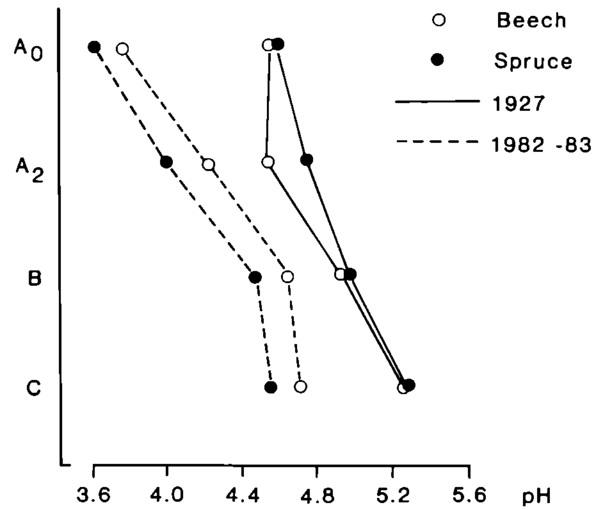
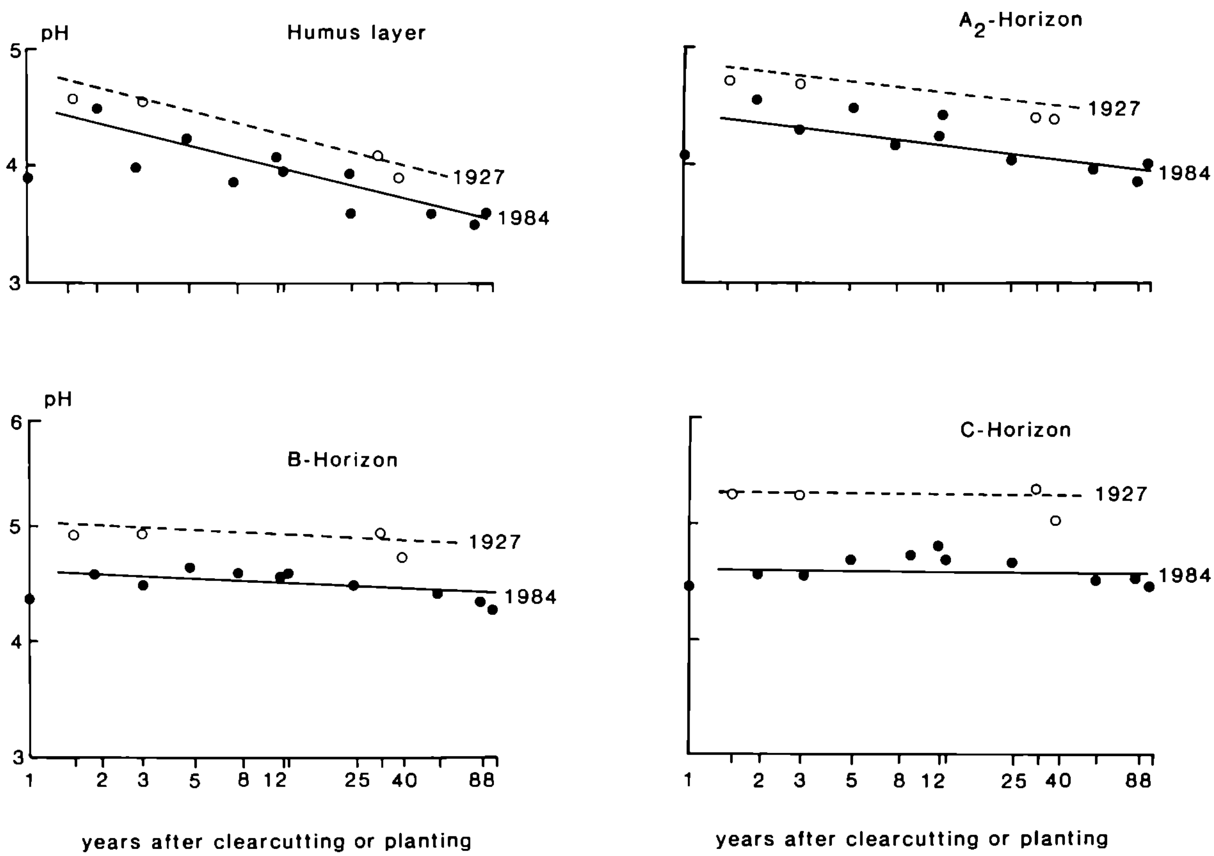


Figure 8. pH of field-moist soil plotted against the logarithmic age for different spruce stands at Tönnersjöheden Experimental Forest, Sweden. (Source: Hallbäcken, L. and Tamm, C.O. (1986). Changes in soil acidity from 1927 to 1982-1984 in a forest area of south-west Sweden. *Scandinavian Journal of Forest Research* **1**, 219-232).



The measurement sites included beech, oak and spruce woodland of various ages. Both stand age and species were found to affect the extent of the increase in acidification, which ranged from 0.3 to 0.9 pH units. The increases in the acidity of the profiles are shown in Figure 7. The pH decrease found in the soils of a forest in southern Sweden (Tönnersjöheden) was not matched by a similar decrease in the soil from a northern forest (Kulbäcksliden). Long-term increases in the acidity of soils in forests have also been reported from Austria[73,80] and West Germany[84,150].

There are several ways by which a soil can become acidified. In the example shown in Figure 7, the increased acidity was explained in terms of biological acidification enhanced by the effects of acid deposition. Biological acidification occurs as a result of the growth and harvesting of trees and may be important in some soils. As a tree grows, it absorbs cations (particularly Ca^{2+} and Mg^{2+}) through the roots. To maintain electrochemical neutrality within the soil solution, hydrogen ions (H^+) are released, and these are the cause of the acidification. The effect decreases with increasing depth in the soil (Figure 8). Within humid temperate areas, natural acidification may also occur as a result of leaching by carbonic acid formed in the soil. This process is dependent on pH and is only important when the soil pH is greater than 5.

Acidification may also occur as the result of man's activities. In some areas, the addition of fertilisers is very important. Application of nitrate will not normally cause acidification as the uptake of NO_3^- is balanced by the release of other anions (usually hydroxyl or hydrogen carbonate). Fertilisers containing cations, such as Ca^{2+}, Mg^{2+} or K^+, may result in acidification if hydrogen ions are released to compensate for the uptake of the nutrients. Ammonium may be particularly important in acidification as it can both be taken up directly by plants, resulting in the release of H^+, or it can generate H^+ when it is broken down in the soil to nitrate. If some of that nitrate is then leached, the acidifying effect will be even greater[134].

Inputs of atmospheric acidity may also be important. The majority of atmospheric acidity is associated with sulphate and nitrate anions. The other major atmospheric anion, chloride, is normally associated with the sodium cation. Much work has been done on the movement of these anions within forest ecosystems. Chloride tends to be fairly conservative within the system, passing through with relatively little alteration. However, the sodium ions associated with chloride can be adsorbed within soils, releasing other cations. This can be the cause of acidic flushes in streams, although the process is thought to occur relatively infrequently in Britain[79].

The majority of the deposited NO_3^- remains within the system because of uptake by tree roots[1]. The movement of sulphate is more complex, but needs to be understood as it is much more important than nitrate in relation to soil acidity. It appears to be much more mobile than nitrate and, if leached from the soil, it may take significant quantities of cations with it[98]. This may be important if the cations involved are nutrients (e.g. Ca^{2+}, Mg^{2+}) or if they are potentially toxic metals (e.g. Al^{3+}). However, research in the United States has shown that only a small proportion of the sulphates deposited within an ecosystem reach the streams[177]. When the soil becomes saturated with sulphates, a balance may be reached between sulphates entering and leaving the ecosystem. This stage appears to have been reached in some forested areas in Britain[127].

The existence of such a balance is dependent both on the amount of sulphate entering the soil and the concentration of sulphuric acid within the soil. The ability of the soil to absorb sulphate from the soil solution depends upon its inherent adsorption properties (determined by the chemical composition of the soil), the amount of sulphate that the soil has already adsorbed and the concentration of the sulphate in the soil solution relative to the concentration to which the soil has previously equilibriated[99]. This means that as sulphate additions increase, increasing amounts can be adsorbed by the soil until a balance between the solution and soil concentrations is reached[180]. Thus, when sulphate in a solution of pH 3.8 was added to a mineral soil, 94 per cent was adsorbed. Only 52 per cent was adsorbed at pH 4.8 and 13 per cent at pH 6.3[131]. Similar results have been obtained from a rendzina[110]. When the pH of the soil solution falls below about pH 4.0, dissolution of aluminium occurs and the ability of the soil to adsorb sulphate decreases. If the concentration of sulphate within the soil water decreases, sulphate will be desorbed from the soil, resulting in leaching of sulphate from the system. This balance has important implications for soil acidification as it indicates that the total volume of hydrogen ions deposited is less important than their concentration. Experimental work has indicated that the maximum removal of sulphates in mineral soil occurs at approximately pH 4.0. Soil chemistry is also important as poorly-buffered soils adsorb relatively little sulphate, whereas soils rich in amorphous aluminium or iron oxides but relatively low in organic matter can adsorb significant amounts[36,55].

Three different things can happen to the hydrogen ions entering the soil from precipitation or throughfall (rain passing through vegetation before it reaches the ground)[120]. They can be neutralized by chemical reactions within the soil, they can be leached from the upper horizons into deeper horizons or groundwater or removed in streamwater, or they can be permanently taken up within the soil, a process which may result in the loss of any mobile basic cations (such as Mg^{2+} and Ca^{2+}) present in the soil. Uptake by vegetation is not included here as the H^+ will ultimately be returned to the soil unless the vegetation is harvested. Leaching of basic cations can result in a change in the soil's acidity, leading to permanent changes in the soil which could be harmful through effects such as the mobilization of toxic elements (e.g. aluminium).

The effects of acidification on soil processes are disputed. This is hardly surprising, given the enormous variation between soils. Experimental work, using pH values commonly encountered in the field (pH 5.7 to 3.5) has indicated that an increase in acidity has little effect on decomposition activity within the soil[40] although a change in the fungal community may occur[11]. However, increased acidity may affect the balance of the soil's natural acids, resulting in the selective removal of some components.

As part of a project at Loch Fleet in Galloway, the Central Electricity Generating Board has attempted the reclamation of acidic soils by liming. It is hoped that this will in turn reduce the acidity of the loch's waters. Work in Finland and West Germany on the long-term effects of liming mineral soils indicates that the calcium is retained in the surface layer of forest soils for long periods[44,75]. Similar effects have been noted at Loch Fleet. As it is the humus layer that is usually the most acidic (see Figure 7), the retention of calcium in this layer is significant.

Acidification of Streams Draining Forests

It is now generally accepted that there has been an increase in the number of acidified freshwaters in Britain and elsewhere during the last 100 years. The causes of this acidification are still being intensely debated, although the importance of atmospheric pollutants is beyond doubt[171]. The acidification of lakes through time can be assessed by examining the fossil diatoms (minute algae) preserved in the lake muds. The particular species of diatom present in a lake depends on the lake's acidity, and an examination of the fossil diatoms reveals past levels of acidity in the lake. In most cases, acidification has occurred naturally since the last glaciation, albeit at a very slow rate. Over the last 100 years, the rate of acidification of some lakes has increased dramatically. The onset of the increase varies between countries. In Norway, the dates range from 1890 to 1930[43], in Finland and Sweden from about 1960[152,153] and in south-west Scotland from 1850 to 1920[56]. While the majority of pH values have declined less than 1 pH unit, some have been reduced by as much as 1.7 pH units[7].

There is a mounting body of evidence associating an increase in the acidity of streams with upland plantations in Britain[26,78,132,179]. This work indicates that streams draining some forested catchments may be more acidic than streams draining some moorland catchments. There is evidence that streams may have become more acidic as a direct result of forestry operations and, in particular, acidic flushes may occur during the establishment of plantations. However, it is important to remember that catchments under coniferous forest do not necessarily have more acidic streamwater than similar catchments with semi-natural grassland vegetation[154].

There are several possible reasons why the water draining forested catchments should be more acidic than water draining moorland. Evaporation and transpiration may result in losses in the runoff from the forest with the result that the water will have higher concentration of solutes[30,31]. Management practices such as ground preparation and drainage may upset the mineral cycling in the ecosystem, resulting in the release of hydrogen ions following an increase in nitrification. Cultivation and drainage may also reduce the time that water is held within the soil, so that the time available for neutralization is cut short. As stated above, the trees themselves may release hydrogen ions into the soil solution. This acidity will remain in the soil unless mobile anions (e.g. those present in acid rain) are passing through the soil[130]. Trees are believed to be much more efficient at intercepting atmospheric pollutants than grass or moorland vegetation and they may act as a filter, with the pollutants being transferred from the air to the streams draining the forested areas.

Studies of fossil diatoms indicate that the problem of water acidification is extremely complex. In Galloway, lochs without any afforestation have become acidified and lochs in afforested catchments were already acidified before the trees were planted. A number of lochs in non-afforested catchments have become acidified by about 1 pH unit since about 1840[8]. The studies of diatoms also indicate that soil acidification and freshwater acidification may not be related. Data from the Round Loch in Galloway[102] suggest that the replacement of mixed deciduous woodland by acidic peatland (dominated by heather, sedge grasses and *Sphagnum* moss) was not accompanied by any change in the acidity of the surface waters in the catchment. In addition, the data indicate that the acidification of soil is insufficient as a mechanism to account for the substantial increase in acidity (about 1 pH unit) of the Round Loch over the past 100 years.

The Forestry Commission has implemented various techniques aimed at reducing the possible impact of forest drainage on acidification. This follows evidence that the runoff from storms may be particularly acidic[35,118]. The most important techniques are aimed at reducing the rate of runoff to streams, thereby increasing the time available for the acids in the water to be neutralized. This includes stopping drains before they reach streams and leaving stream margins unplanted.

Forestry operations prior to planting may also cause the acidification of streams, as ploughing and draining may disrupt the nutrient cycle in the soil. In many organic soils, ploughing is accompanied by increased oxidation and mineralization, with nitrates, ammonium, phosphate and sulphate being produced[88]. As the vegetation is poorly developed at this stage, there may be severe leaching of these compounds. Drainage is accompanied by a substantial increase in the rate of runoff, and the reduced soil-water residence times may also contribute to the acidity of the drainage water. Further problems may occur at this stage in the afforestation cycle as a result of increased sediment in streams.

In general, the acidification that normally occurs with the growth of trees is thought to be slow and unlikely to lead to the acidification of streams and lakes[130], although this requires experimental verification. Consequently, it seems likely that the increased acidification of streams draining some afforested areas is a direct consequence of increased levels of acidic deposition. This conclusion is supported by the finding that increases in the concentrations in some loch sediments of trace metals only found in man-made emissions are correlated with the increase in acidity[8].

Symptoms of Forest Damage

A wide variety of symptoms have been recognised in affected trees. There is considerable variation between species and there are also regional variations in the nature of the damage within a given species. This presents problems in making comparisons. The basic variables used in the national surveys of forest health are the extent of needle and leaf loss and the degree of discoloration of the leaves and needles (Plates 1 and 2). Needle and leaf loss is either used on its own or is used

Table 2. Classification of damage type (as agreed by the International Cooperative programme on the assessment of air pollution effects on forests).

Needle/leaf loss only

Class	Needle/leaf loss (%)	Tree health
0	0-10	Healthy
1	11-25	Slight damage
2	26-60	Medium to serious damage
3	61-99	Dying
4	100	Dead

Needle/leaf loss and discoloration

Damage class	Degree of yellowing (% of yellowed needles/leaves) 0-25%	26-60%	61-100%
0	0	1	2
1	1	2	2
2	2	3	3
3	3	3	3

in combination with needle and leaf colour (Table 2). Problems exist with these indices as they may be induced by a variety of factors, of which air pollution is only one.

A further complication is that some agents, including air pollution, are thought to be capable of causing 'latent injury'. In such cases, damage exists, but it is not readily apparent. For example, beech seedlings that were fumigated for an entire winter with 112 ppb SO_2 showed a reduction in the numbers of terminal buds that developed in the following spring[106]. Similar delayed responses have been found in fumigation experiments of Scots pine[69]. Consequently, the interpretation of the symptoms that have been observed is extremely difficult.

Forest Damage in Britain and Abroad

In the 1970s, it became apparent that forests in several high-altitude areas in West Germany were showing signs of dieback (Plate 3). At first, the damage was restricted to Silver fir, but by 1978, Norway spruce was also showing signs of decline. In the period 1980 to 1984, damage was recognised in an increasing number of species and areas, both in Germany and elsewhere in Europe and in eastern North America. The damage appears to be unprecedented and is frequently referred to in the German language literature as the 'neuartige

Plate 1. Norway spruce showing many of the symptoms typically associated with air pollution damage. In particular, note the marked yellowing of older needles and the defoliation in the area below the crown. (Bavaria, summer 1986). *(37579)*

Plate 2. Needle chlorosis in Norway spruce. *(37580)*

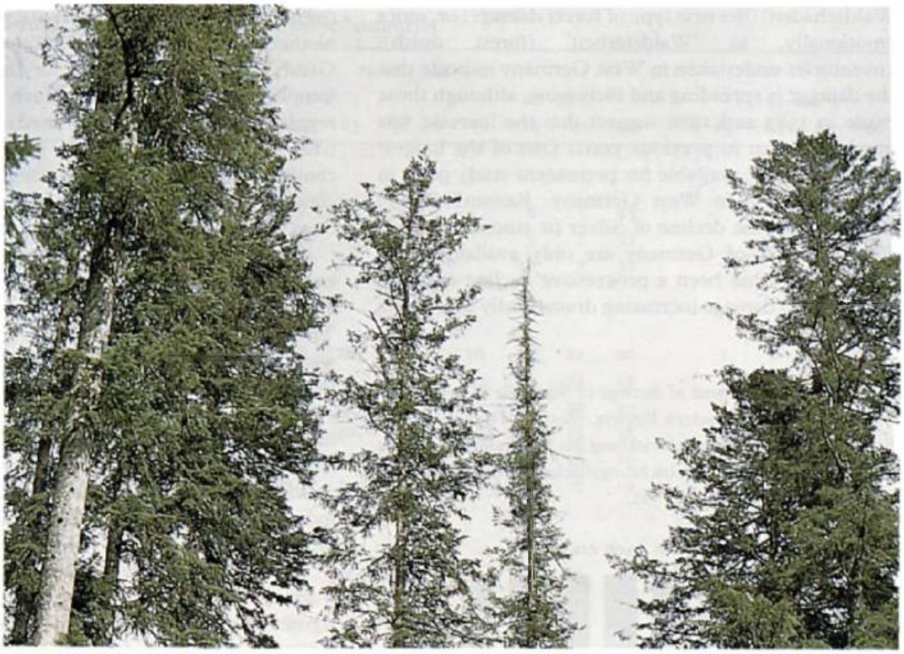
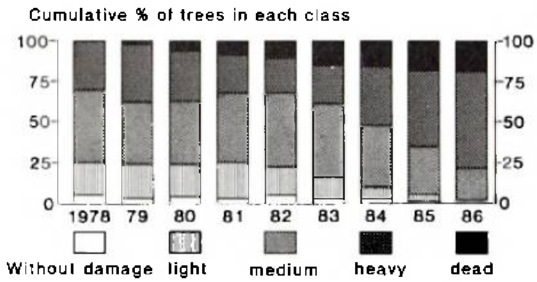


Plate 3. Severely damaged Silver fir and Norway spruce in the Bavarian National Park (summer 1986). *(37582)*

Plate 4. Norway spruce stand at Luchsplatzl in the Bavarian Forest (summer 1986). The stand showed signs of decline in the early 1980s and has now been badly affected by bark beetles (*Ips typographus* and *I. chalcographus*). The tree deaths have probably been induced by the bark beetles, although damage occurred prior to their attack. Normally, stands such as this would be felled to discourage bark beetle attack. *(37581)*

Waldschäden' (the new type of forest damage) or, more emotionally, as 'Waldsterben' (forest death). Inventories undertaken in West Germany indicate that the damage is spreading and increasing, although those made in 1985 and 1986 suggest that the increase was much less than in previous years. One of the longest data sets is that available for permanent study plots in eastern Bavaria in West Germany. Research here[89] demonstrates the decline of Silver fir since 1978 (data for the whole of Germany are only available since 1983). There has been a progressive decline over the period, with damage increasing dramatically since 1982 (Figure 9).

Figure 9. Development of damage to Silver fir in permanent observation plots in eastern Bavaria. (Source: Hörteis, J. and Schmidt, A. (1986). Die Entwicklung des Gesundheitszustandes der Weisstanne auf 10 Beobachtungsflächen in Ostbayern. *Der Forst und Holzwirt* **41**, 580-582).



As with all systems of classification, there are problems with that adopted for the surveys of forest health (Table 2). For example, most forest scientists now feel that up to 20 per cent needle loss in conifers may occur without implying 'damage', and the lack of an absolute baseline is probably the single greatest problem in the assessment of the impact of air pollution on forest health. In addition, tree-ring analysis has suggested that visible damage may only be the final symptom in a long history of decline[4,168]. However, the adoption of the classification shown in Table 2 means that at least the condition of sample trees can be compared between countries.

The forest damage inventories undertaken in Britain and elsewhere record the extent of damage to trees in sample plots. Figures are normally given as the percentage of sample trees in a given damage class. In some countries, these may be representative of the overall state of the forests (e.g. West Germany, Netherlands, Luxembourg), in others they are not (e.g. Great Britain). The reason for this is related to the sampling design. In countries such as West Germany, a regular grid (4 km x 4 km) is used, and a set number of trees are sampled at each grid intersection. The combined data from all the sample stands are then considered to be indicative of the total population of trees in the country.

In Britain, the principal aims of the survey are to establish whether or not damage is present and to relate any damage found to specific causes as far as is possible. The sampling design is therefore aimed at covering as wide a range of conditions as possible. Consequently, the plots are not representative of the total population of trees in the country, and the data must not be interpreted as being so. In both approaches, only a small proportion of the trees in a stand are assessed (usually 24). At present, there is little information to indicate whether a sample of this size is representative of the stand.

The results for a number of countries are given in Table 3. Care should be taken in interpreting these figures because of the differences in assessment techniques between different countries and, in some cases, differences in the assessments through time in individual countries. In West Germany, it is apparent that damage increased substantially between 1983 and 1984, but that the rate of increase was less in 1985 and 1986. In Switzerland and the Netherlands, damage has increased through the period 1984 to 1986. In Great Britain, it now (1986) appears that the condition of the trees examined in the survey can only be classed as moderate[92].

Damage is not restricted to Europe. Forest decline is occurring in North America. Air pollution is known to have affected trees in some areas, particularly in California where ozone and other photochemical oxidants are important. Ozone has also been identified as one of the primary causes of the ill health of Eastern white pine (*Pinus strobus*) in the east of the USA. In the 1980s, a widespread and severe decline of Red spruce (*Picea rubens*) and, to a lesser extent, Pitch pine (*Pinus rigida*) and Shortleaf pine (*Pinus echinata*) has been noted[96,97]. The decline in some areas bears striking similarities to that occurring in parts of Europe in that there is a progressive loss of foliage, invasion by secondary pathogens and, in many cases, mortality.

Severe dieback of Japanese red cedar (*Cryptomeria japonica*) has been reported from Japan where it is believed to be associated with damage by oxidants[172].

Table 3. Forest damage assessment in five European countries.

	Needle/leaf loss															
	0-10%				11-25%				26-60%				61-100%			
	83	84	85	86	83	84	85	86	83	84	85	86	83	84	85	86
United Kingdom																
Sitka spruce		65	83	45		28	12	39		6	5	15		1	0	1
Norway spruce		71	84	32		26	15	36		3	1	31		1	0	1
Scots pine		49	74	25		29	18	41		16	7	32		5	1	3
West Germany																
Norway spruce	59	49	48	46	30	31	28	32	10	19	21	20	1	2	3	2
Pine	56	41	43	46	32	38	41	40	10	20	15	13	1	1	2	1
Silver fir	25	13	13	18	27	29	21	22	41	45	50	49	8	13	16	11
Beech	74	50	46	40	22	39	40	41	4	11	13	18	0	1	1	1
Oak	85	57	45	39	13	35	39	41	2	9	16	19	0	0	1	1
Other trees	83	69	69	65	9	24	23	25	8	7	7	9	0	1	1	1
Switzerland																
Norway spruce		65	63	50		28	29	36		6	6	12		1	2	2
Pine		50	35	34		31	47	43		16	13	19		1	5	4
Silver fir		62	60	47		27	28	36		9	8	13		2	4	4
Larch		64	66	39		28	23	44		7	7	12		1	4	5
Beech		74	69	52		23	27	40		3	3	7		0	1	1
Oak		71	60	37		28	33	50		1	6	11		0	1	2
Maple		86	86	73		11	11	25		2	1	1		1	2	1
Ash		84	77	57		13	20	36		3	2	7		0	1	7
Netherlands																
Norway spruce		62	48	49		28	41	34		7	9	12		3	2	4
Scots pine		34	48	50		51	36	33		12	14	13		2	2	3
Corsican pine		57	40	19		34	42	29		8	15	40		1	3	12
Douglas fir		50	33	17		39	43	27		9	22	45		2	2	11
Beech		71	72	68		24	21	26		4	6	5		1	1	2
Oak		57	40	30		38	39	42		5	19	20		1	2	9
Luxembourg																
Norway spruce		79	84	87		17	12	10		3	3	2		2	1	1
Oak		59	77	81		34	20	16		6	3	2		2	1	0
Beech		66	70	67		28	28	27		5	5	6		1	1	1

Possible Causes of Forest Damage

Many hypotheses have been put forward in an attempt to explain the observed dieback of trees in Europe and North America. In some cases, the decline is attributed to a single factor, in other cases a combination of factors is believed to be involved. Some of these hypotheses invoke air pollution as a primary or secondary factor, others do not. Several commonly attributed factors are discussed below.

Forest management practices

It has been argued that much of the decline can be attributed to the effects of changing from broadleaves to conifers over the past few centuries[37]. It is suggested that the change from deep-rooting broadleaves to shallow-rooting conifers has led to the compaction of the deeper layers of the soil, thereby increasing the susceptibility of trees to drought. When combined with the increased acidification associated with conifers, these semi-natural processes could lead to a marked increase in the susceptibility of trees to other stresses. While this argument may help to explain the decline of some managed stands of conifers, it fails to explain either the decline in areas where there have always been conifers or the decline of broadleaf forests. Furthermore, the change from broadleaves to conifers occurred over several centuries, yet whereas some symptoms of forest decline have occurred in the past, the widespread and severe decline observed today appears to have developed within the last 40 years.

An attempt has been made to evaluate the role of management in forest damage in West Germany[27]. The study concluded that although managed forests were more susceptible than unmanaged forests to large-scale damage by wind, snow and insects, there was no evidence of a link with forest damage. This finding is consistent with the results of the 1984 West German forest damage survey. Similarly, no evidence has been found of a link between management and beech health in Switzerland[58]. Management practices can therefore be ruled out as a primary cause of the decline currently observed over much of Europe.

It is important to remember in this context that, as tree stands age, there is a considerable amount of mortality as a result of competition for light, nutrients and water. This occurs in both natural and managed stands. Consequently, suppression must be taken into account when examining the health of individual trees within a stand.

Long-term climatic change

The evidence for the effects of long-term climatic change on tree health is limited. Long-term climatic trends have occurred in central Europe over the last 100 to 200 years and although they are poorly defined, two periods of markedly increased temperatures (1890-1900 and 1940-1950) are apparent[165]. Analysis of long-term records in the form of mean monthly precipitation or temperature records is unlikely to yield information of value to the present problem as extreme values are normally of more importance to tree health (see below). However, reconstructions of monthly soil moisture deficits may be useful. It is possible that factors such as a change in the frequency of frosts or in the amount and frequency of precipitation are implicated but the impact of such trends is very hard to evaluate. The general opinion is that long-term climatic effects have had little impact on tree health up to the present time.

Extreme climatic conditions

It has been suggested that drought triggers forest damage, and past periods of decline of both Silver fir and Norway spruce have been associated with low rainfall in the growing season[38]. However, that study assumed that the present forest decline in Europe is no different from declines that have occurred in the past. Most forest scientists disagree with this, the present decline being considered much more severe and widespread. In Europe, the droughts of 1976, 1983 and 1984 caused severe stress to many species of trees, particularly beech. Drought has also been implicated in the decline of Red spruce in North America[96] and it is thought to interfere with root regeneration, resulting in the loss of efficiency of water uptake[140] and drought-associated root damage may enable invasion by fungal pathogens. Drought may also indirectly affect the levels of soluble sugars in the bark and cambium (as a result of defoliation and subsequent reflushing) which may predispose the trees to fungal or insect infestations[192].

Frost and winter cold have also been associated with forest decline. In particular, magnesium deficiencies either occurring naturally or induced by air pollution may result in the reduction of the frost-hardiness of trees. Consequently, when severe frosts occur (as they did in parts of West Germany in 1979, 1981, 1982 and 1983), severe damage or even death may result[142]. Damage to trees in Britain has been associated with sudden fluctuations in temperature, with typical symptoms being the death of needles and shoots during the dormant season. Winter damage may also occur when freezing conditions are accompanied by strong winds, resulting in the desiccation of conifers. This was particularly marked in some areas of Britain in early 1986, and it is believed to have been a major factor in the increase of damage reported in Britain in 1986[92]. The possibility that some other factor predisposes trees to damage by both frost and winter cold processes cannot be ignored.

Gaseous pollutants

Many gaseous pollutants are know to injure trees. Some of these are generated from a single source and have only local effects, others may be derived from numerous sources or have effects over large areas or both. Of the gases that affect wide areas, three are generally considered to be important. These are sulphur dioxide, the nitrogen oxides and ozone. Other gases (e.g. peroxyacetylnitrate) do occur, but it is not known whether, in Britain, these reach concentrations likely to damage trees. This account concentrates on SO_2, NO_x and O_3. It is important to distinguish at the outset between acute and chronic effects. Acute effects occur as the result of a single episode, when concentrations are sufficiently high to damage the plant tissues. Chronic effects, which also involve damage to the plant tissues, occur over longer time periods and are caused by lower concentrations. Chronic exposures are most likely to result in severe stress to the plant; acute exposures may kill the plant. The distribution of chronic pollution through time is important. If the pollution is episodic, as is frequently the case, then the plants may be able to recover between occurrences of high concentrations, as has been shown for aspen and Jack pine in Canada[85].

SO_2 is known to damage plants, but experimental work using fumigation chambers has suggested that the concentrations that are required to induce visible foliar damage to common forest trees rarely occur in Britain at the present time. However, the concentrations that do occur may affect growth when combined with other pollutants. Trees may be indirectly affected by SO_2 through soil-mediated effects, and experimental work on the effects of dry deposition of SO_2 on spruces has shown that the mineral supply (particularly of calcium and magnesium) may be adversely affected[103].

The mean annual SO_2 concentrations recorded in the Black Forest and in some parts of Bavaria are generally less than 40 µg m^{-3}, well below the concentrations required to damage trees. Elsewhere in Germany, notably the Fichtelgebirge of north-east Bavaria, SO_2 concentrations are known to reach levels that are likely to injure plants and the gas is probably involved in the damage in such areas. In central Hesse, daily average SO_2 concentrations during a 7-day period in January 1985 were 800 µg^{-3} with peak levels of 1600 µg m^{-3} [194]. This episode may well have injured trees in the area.

A major argument against SO_2 as a pervasive cause of the forest decline is the presence of damaged trees in areas with low SO_2 levels. For example, a particularly badly affected area in West Germany occurs around Freiburg, where the annual mean SO_2 concentration is only 12 µg m^{-3}. However, peak hourly values may be much greater. The differences between mean annual values (which are the values that are commonly shown on maps) and peak values (which may be the values that actually cause the damage) makes it very difficult to compare regional levels of pollution. The relationship between chronic and acute damage is extremely hard to evaluate, and generalisations are probably impossible[59]. However, damage by SO_2 certainly occurs in some areas (such as parts of eastern Europe), although its extent is probably fairly small in relation to the total area now being affected by forest decline.

Recorded levels of NO_2 are generally well below those known to cause damage to trees, and the gas can be ruled out as a primary cause of the damage. However, it should be remembered than NO_2 is an important precursor for ozone.

Ozone is currently thought to be involved in the dieback observed in several countries (notably Germany and the USA). Concentrations of O_3 are known to reach phytotoxic levels over much of Europe, and an examination of the microscopic damage to Norway spruce needles in the Black Forest has shown symptoms typical of damage by ozone[19]. High concentrations of O_3 occur more frequently at higher altitudes, where damage to trees was first observed and is still most severe. However, the spread of damage to lower altitudes and to relatively O_3-tolerant species such as beech and oak suggests that the effects of O_3 alone are insufficient to explain the occurrence of the dieback everywhere.

An increasing number of experiments are showing that growth of both conifers and broadleaves is significantly greater in charcoal-filtered air (i.e. with low SO_2 and O_3) than it is in ambient air. There has been some important work on the interaction of O_3 with other pollutants, particularly 'acidic mist'[18,39,105]. It is believed that O_3 affects the permeability of individual cell walls, easing the removal of ions from the leaves by acidic rain and mists. The effect of the interaction between O_3 and acidic mist is still controversial as some work in Britain[23,175,176] and the United States[47,147] has failed to reproduce the German results exactly. Furthermore, the symptoms produced in laboratory experiments by this combination of pollutants are different from those found in the field[114]. Recent work[113] suggests that the rate of leaching of ions depends on several factors, including seedling vigour, soil nutrient content and the pH of the mist. This may be the reason for the apparent differences in the results obtained by different workers.

Some pollutants may interact with climate to cause damage. In particular, SO_2, O_3 and NH_3 are all believed to reduce the frost hardiness of conifers, although by different mechanisms[23,54,108]. Given that SO_2 and NH_3 emissions are likely to be highest in periods when frost is most likely, the reduction in frost-hardiness may be significant in some areas where high concentrations of these gases occur. Increases in frost sensitivity are considered by some to have been the initial factor leading to forest decline.

A major problem with much of the work on the direct effects of gaseous pollutants is that many studies deal with single gases or unrealistic concentrations. Recently, there has been increasing interest in the simulation of realistic conditions, using relatively low concentrations of a combination of pollutants. Existing studies indicate that some pollutants may work together, with trees being more sensitive to mixes of gases than to individual pollutants. For example, Norway spruce seedlings are known to be much less sensitive to individual gases than to combinations of O_3, SO_2 and NO_x [76]. Many of the results from such experiments are difficult to interpret because of differences in experimental technique. A further difficulty is that there is now evidence that there may be rapid selection within a population of trees towards pollution-tolerant individuals[13]. This will have considerable effects on the application of experimental results to trees growing in polluted environments.

Acidic rain

The possibility that acidic mists, acting in combination with gaseous pollutants, damage trees has already been discussed. Similar findings have been reported for interactions between acidic rain and gaseous pollutants[33,39]. There have also been suggestions that acidic mist and rain could directly damage trees. At present, there is no evidence that rainfall less acidic than pH 3.0 can damage the foliage of mature trees[126] although there is evidence of increased mortality when seedlings of some species are treated with precipitation at pH < 4.0[133]. In some species, it is possible that irrigation with acidic precipitation containing sulphate and nitrate could induce increased growth as a result of the increased availability of nutrients[123]. However, there are other possible mechanisms by which acidic rainfall could damage trees. In particular, stemflow acidities can be very high and may cause marked acidification of the soil at the bases of trees[57].

Experimental work has suggested that acidic precipitation increases the rate of removal of nutrients from the canopy. The problem here is in establishing whether the nutrients are being removed from within the foliage or whether they are derived from the solution of nutrients already on the foliage surface[109]. At the present time, this question has not been answered.

Soil acidification

Acidification of the soil, with consequent increases in the concentrations of phytotoxic elements such as aluminium, has been argued as the cause of forest decline in West Germany and elsewhere[188,189,190]. The hypothesis has been questioned on the grounds that damage occurs on a wide variety of soil types, some of which are well-buffered against acidification. However, soil acidification may be important for certain species: beech roots have been found to be much more restricted in acidic soils than in base-rich soils[72]. Beech is known to favour less acidic soils and it is thought that the better supply of nitrogen, phosphorus, calcium and magnesium and the absence of toxic levels of aluminium and manganese may be responsible[144]. In acidified soils, the roots of some trees such as beech are restricted to the A horizon of the soil and any that penetrate lower down show signs of injury. Experimental work suggests that the presence of aluminium in these acidic conditions may be important, although some scientists have failed to find any evidence that trees are suffering from aluminium toxicity[97,161]. The increase in stream aluminium levels that has occurred in acidified areas suggests that aluminium may be readily leached from acidic soils, indicating a possible method by which concentrations of aluminium in the soil may be kept comparatively low.

The dynamics of aluminium within a forest ecosystem in the United States have recently been examined[186]. Concentrations of aluminium in various parts of the ecosystem could be directly related to the movement of water through the soil, the concentration of dissolved organic carbon and the pH. The concentration of aluminium in precipitation averaged at 0.012 mg l^{-1}. This increased to 0.14 mg l^{-1} in throughfall and 0.45 mg l^{-1} in the lowermost horizon of the soil. Much of this remained within the soil, and the mean concentration in the stream was only 0.14 mg l^{-1}. Large variations in the concentrations of aluminium in the soil were found, and the measured concentrations also varied according to the type of instrument used to record them. The study indicates that much more research is required before the role of aluminium released by increased acidity can be evaluated.

As already intimated, the importance of elevated aluminium levels as a direct consequence of soil acidification in the recent forest decline is unclear. There are three ways in which aluminium could affect plant growth[159]. Firstly, there could be a reduction in the uptake of divalent cations as a result of the presence of aluminium in the root tissue. Secondly, aluminium could cause a breakdown in the normal growth and functioning of the root cells. Thirdly, increased levels of aluminium in the immediate vicinity of the roots could result in an increase in the number of anion adsorption sites, thereby reducing their availability to plants. The interactions with aluminium are extremely complex but, in general, it appears that conifers are well able to withstand the levels of aluminium commonly encountered in acidic soils[119,159,160].

Excess nutrients from the atmosphere

This phenomenon is thought to be important in Sweden and the Netherlands, although it may well be significant in other areas with high emissions of ammonia or high rates of nitrogen deposition. The basic argument is that increased deposition of nitrogen is leading to tree damage. The increases may be the result of nitrates derived from interactions between nitric oxides and the atmosphere, or they may be the result of emissions of ammonia.

Ammonia is released from a variety of sources, the most important in Europe being farm slurry. About 20 per cent of the ammonia that is emitted is deposited within 5 km of the source, the remainder being converted into ammonium compounds. The ammonium/ammonia can have a variety of effects on the soil. These include: i) the acidification of the soil when the ammonia is converted (via ammonium) into nitrate[20], ii) the supplanting of some important nutrient ions such as Mg^{2+}, K^{+} and Ca^{2+} by ammonium, leading to shortages of these ions[157], iii) the enrichment (eutrophication) of nutrient-poor regions, and iv) the encouragement of acidification by enhanced deposition of SO_2, which arises as the result of the rate of deposition of SO_2 being greater on needles that have a less acidic surface because of the presence of the ammonium.

The acidification of the soil as a result of ammonia and ammonium deposition appears to be a self-regulating process in that nitrification stops or at least becomes severely inhibited at pH levels of less than 4.1. In parts of the Netherlands, soils have reached this level and do not appear to be becoming more acidic[157]. The impact of ammonium seems to be related to the presence of other nutrients in the soil, and it has been suggested[3] that trees on soils that are poor in nutrients will not grow well if the NH_4^+/K^+ ratio in the soil is greater than five.

Nitrogen may also be supplied as nitrate or directly taken up in the form of NO_2. Nitrogen dioxide can either be beneficial or toxic to plants, depending on the dosage that is received. At relatively low concentrations, it provides an extra source of nitrogen. As the concentration increases, the beneficial effects are out-weighed by the toxic effects. In this context, it is important to place atmospheric inputs of nitrogen into perspective. In Britain, the application rate, when used, is normally 150 kg N ha^{-1} (applied as 350 kg urea ha^{-1}). Applications of N may be necessary every 3-4 years on some peaty soils. (N.B. Nitrogen fertilisation is relatively rare in Britain because of the cost and the need for frequent applications). Atmospheric deposition of nitrogen in Europe varies from about 10 kg N ha^{-1} yr^{-1} to over 60 kg N ha^{-1} yr^{-1}. Nitrogen fixation by plants can supply even greater amounts of nitrate, with fixation rates of 50 to 200 kg N ha^{-1} being reported for Red alder in the western United States[124].

Although the increase in the amount of nitrates available to plants may improve growth as a result of fertilisation, problems may arise. These occur when nitrates are made available in autumn, as the carbohydrates that are required to supply the energy for the breakdown of the nitrates into a form suitable for use by the tree may not be replaced by photosynthesis. As a result the winter hardening of the tree may be adversely affected. This is thought to be of importance in the decline of Red spruce in some parts of North America[24]. In addition, the rise in the assimilation of nitrogen requires increased supplies of magnesium, potassium, phosphorus, molybdenum, boron and water[129], and deficiencies in these may lead to stress. In particular, an abundance of nitrogen may cause a reduction in the growth of roots relative to the growth of shoots, resulting in a reduction in phosphate uptake relative to nitrogen[125]. This may lead to phosphate deficiency. Increased levels of nitrogen are also known to inhibit root mycorrhizas[173] and increase the susceptibility of the roots to fungal infections such as *Pythium, Rhizoctonia* and *Phytophthora*[90].

There is little evidence that excessive levels of sulphur are involved in the forest decline. However, it has been argued that levels of sulphur in Norway spruce needles at a site in Belgium may be toxic[135]. If good growth is to be maintained, nitrogen to sulphur ratios in foliage should remain below eight[196]. More work in this field is required.

Nutrient deficiencies

Nutrient deficiencies can occur naturally as a result of the geology of an area. However, pollution may cause additional problems. As mentioned above, damage by pollution may lead to the loss of nutrients from tree leaves and needles. The acidification of the soil may also lead to increased loss of nutrients through leaching[81,187]. The latter process is more important as the nutrients are being lost from the ecosystem, whereas, in the case of foliar leaching, the nutrients enter the soil and may be taken up again by the plant. At present, there is only limited evidence that increased foliar leaching is accompanied by a reduction in the nutrient content of the foliage[58], although accelerated leaching may be accompanied by foliar deficiencies of magnesium and calcium if the soils are poor in these nutrients[18]. Uptake by plants will be impeded in cases where the roots are damaged, and there is some evidence that soil acidification may be accompanied by such damage[122]. In addition, ozone may interfere indirectly with root microbial associations [148], although the implications of this are yet to be fully assessed. Root injury has been found on a number of damaged spruces in West Germany, and may lead to nutrient deficiencies[178].

Deficiencies of magnesium have been reported in trees showing evidence of decline. Chlorotic spruce needles have been found to have much lower (20-70 per cent) magnesium levels than healthy-looking needles[196]. Magnesium deficiencies have been recorded on acidic soils at higher altitudes (*c.*700-1000 m) in West Germany[166] and foliar magnesium deficiencies are widespread in other parts of Germany[139] (Plate 2). A threshold appears to exist at approximately 0.3 mg g^{-1}, below which needle chlorosis occurs. As magnesium-deficient needles are associated with a breakdown in the basic physiology of the tree, accelerated foliar leaching of magnesium and calcium has been suggested as a possible cause of the forest decline in some parts of Europe[17,141]. Deficiencies in both of these nutrients may predispose trees to damage by photochemical oxidants[76,199].

Other nutrients may also be important. Cadmium, lead, copper, zinc and manganese may all be leached from needles and leaves following damage by air pollution[74]. Preliminary work in Hesse, West Germany, indicates that the manganese content of Norway spruce needles is negatively correlated with the percentage of crown defoliation, suggesting a link between the two[70]. However, more work is required before any cause-effect relationships can be determined.

Uptake of ammonium is normally accompanied by the release of potassium, magnesium and calcium[157]. As these may then be leached from the soil, deficiencies may arise, leading to stress in the plants.

The frequent association of symptoms of forest decline with nutrient deficiencies suggests that some sort of relationship is involved. Following the finding that the decline was occurring on calcareous soils, the nutrient deficiency hypothesis lost favour. However, the calcareous soils that are particularly affected are often poor and the availability of nutrients may be rather low[163,199]. The recovery (both reductions in discoloration and increases in the numbers of needles retained on the tree) of many plots following fertilisation[91,93] is also indicative that problems associated with nutrient deficiencies may be involved with the forest decline.

Trace-metal accumulation

The effects of increased concentrations of potentially toxic metals such as aluminium have already been noted. This section is concerned with the possibility that the atmospheric deposition of heavy metals may have an adverse effect on tree growth. Lead has been proposed as a possible cause of damage to trees in West Germany[51,52]. The lead in question is trialkyl-lead which is formed in the atmosphere from the lead emitted in car exhausts. The theory has been strongly criticised[191] on the basis that the methods used in the determination of the concentrations of lead are subject to a good deal of error. In addition, the concentrations recorded seem much higher than would be expected from a knowledge of the original emissions and subsequent evolution of trialkyl-lead. Differences in the lead levels in foliage on exposed and unexposed trees have been reported from the western slopes of the Egge Mountains in West Germany[5], but much more information is required before the role of lead can be determined.

In general, it appears that the concentrations of heavy metals in damaged trees are extremely low and can therefore be discounted[17,198]. A further argument against heavy metals as a primary cause of the dieback is the lack of damage in areas where pollution by heavy metals might be expected, such as around the refineries in the Stolberg and Rhineland areas of West Germany[137]. There is no doubt that heavy metals can be a problem locally and it has been argued that the poor growth of Sitka spruce in some parts of south Wales could be because of toxic levels of heavy metals[28]. However, there is no evidence that they are involved in the forest decline currently affecting many parts of Europe.

Radioactivity and other radiation

In West Germany, it has been proposed that forest decline can be linked to emissions from nuclear power stations[149]. The hypothesis is based on the tentative observation that the highest degree of forest damage in Baden-Württemberg occurs in the vicinity of the Gundremmingen nuclear power station. The hypothesis can be rejected on the basis of two independent studies. Firstly, in the area around Gundremmingen the damage is not concentrated around the power station[185]. Secondly, there is no relationship between the distribution and extent of damage and the suspected sources of radioactivity[16].

High-frequency electro-magnetic radiation has also been proposed as a posssible factor inducing stress in trees[86]. The energy associated with the electro-magnetic fields created by the radiation may be sufficient to induce relaxation and resonance phenomena in trees and this in turn may result in disturbances to the ion fluxes within needles. Much more experimental work is required before this hypothesis can be assessed.

Viruses

Viruses have been proposed as a possible cause of the decline of conifers in Europe[63,105], but this has not been widely accepted owing to the lack of supporting evidence[128]. The claim rests principally on electron microscope studies which, with present knowledge, cannot be unambiguously interpreted. Viruses have been isolated from declining beech in West Germany and the investigation of their role clearly presents an important, albeit difficult, field of research.

Fungi

It is generally agreed that the forest decline is not primarily being brought about by fungal pathogens. There is, however, considerable discussion on the role that fungi may perform in conjunction with other factors such as air pollution or weather extremes. For example, in Germany there is currently a vigorous debate over the part played by needle-inhabiting fungi in the development of a needle-reddening symptom that has been observed in certain parts of the country. Some have argued for the importance of fungi like *Lophodermium piceae* and *Rhizosphaera kalkhoffii* on trees stressed by 'frost shock'[145,146], while others dispute this and maintain that the fungi are of no real consequence[29,167].

The interaction of air pollution with fungal and bacterial diseases is a matter of obvious interest in connection with forest decline. Air pollution does not necessarily lead to an exacerbation of disease, as the direct effect of the pollutant on the pathogen may be more important than its effect on the plant. Thus tar spot of sycamore is absent from highly polluted areas because of direct inhibition of the causal fungus *Rhytisma acerinum* by sulphur dioxide[14]. In contrast there is evidence that the same pollutant stimulates the spruce needle pathogen *Rhizosphaera kalkhoffii* and increases its damaging effect upon young trees[182]. Many of the claims for a synergistic effect of air pollution and fungal pathogens are inadequately documented[87]. No critical data have as yet been produced for areas of western Europe affected by forest decline but there is evidence for a link between air pollution and increased incidence of root rot fungi like *Heterobasidion annosum (Fomes annosus)* and *Armillaria* spp. in countries such as the United States[95].

One fungal disease that has aroused considerable concern during the mid 1980s is a serious dieback of pine caused by *Sphaeropsis sapinea* in the southern part of the Netherlands[42]. Lines of research include an examination of the effect on host and pathogen of the high levels of ammonia which are known to occur in that part of the country.

Drought is a very important abiotic factor in relation to interactions with fungi and several scientists have drawn attention to the ability of stem and root pathogens to prolong and intensify the poor condition of trees that would have recovered quite quickly from drought stress alone. The situation is even further complicated by the fact that in many places, periods of drought are likely to coincide with the occurrence of high concentrations of pollutants, particularly ozone.

Pests

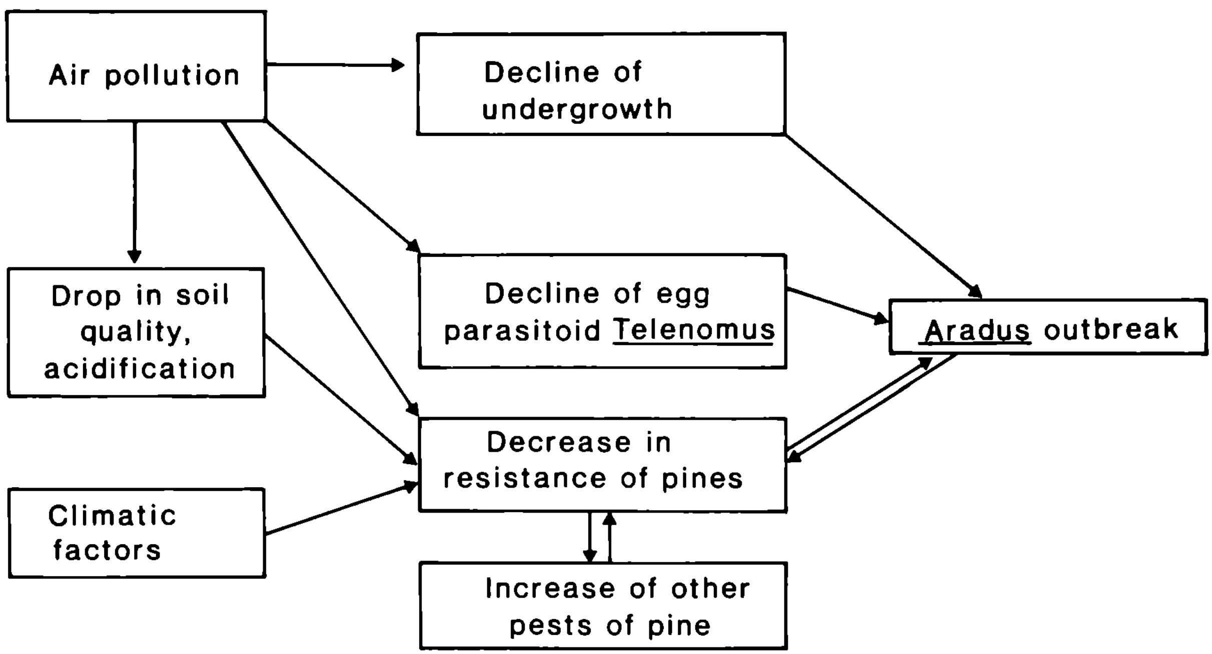
At present, there is no evidence that pests are the cause of the forest damage observed in central Europe. However, they are clearly associated with trees that have been subjected to stress, and it appears that many species have taken advantage of trees damaged by air pollution (Plate 4). As with some fungal attacks, the main problem is in establishing whether insects are the primary cause of a tree's decline or whether they are infesting trees that have already been weakened or injured by some other factor. For example, the Southern pine beetle (*Dendroctonus frontalis*) and the larger 8-toothed spruce bark beetle (*Ips typographus*) both initially attack damaged trees rather than trees in full health[6]. Such attacks may be a specific response to chemicals (such as ethanol and terpenes) produced by trees under physical or toxic stress[71,72]. These may either attract or discourage insect infestations. Volatile terpenes are known to act as a major signal to some species of insects infesting conifers[195] whereas ethanol may either attract some species (such as the ambrosia beetle (*Xyloterus* spp.))[111] or discourage them, as with the cerambycid beetle *Monochamus alternatus*[181]. Once attracted to an area by the presence of unhealthy trees, the insects may start to attack healthy trees and they can then be classed as a primary cause of tree decline.

Several trends are apparent in the dynamics of the populations of forest pests[6]. The frequency of simultaneous outbreaks in various species in Europe has increased from about once every 14 years in the late 18th century to about once every 6 years since 1900[32]. In addition, species that used to be rare now occur sufficiently frequently to be classed as pests[65]. The causes of these trends are unknown, but climatic change, management practices and air pollution have all been suggested.

Insect pests can interact with pollutants in several ways[66]. The quality of the food plant may be changed in such a way as to increase its attractiveness to pests. Pollutants themselves may affect the pests in a number of ways to their advantage. Pollutants may also affect the natural enemies of the pests. Several different responses to pollution exist[66], depending on the level of contamination. Some species such as one of the Tortrix moths (*Exoteleia dodecella*) and the Pine shoot moth (*Rhyacionia buoliana*) may occur in large numbers close to a pollution source[183]. High densities in polluted areas can be achieved by a number of mechanisms, including having high tolerances to ambient pollution levels, avoiding contamination by the pollutants, exploiting badly damaged trees and exploiting environments where their predators have been reduced. Other species peak in areas with moderate pollution, e.g. the bark beetle *Phaenops cyanea* and the 6-toothed pine bark beetle (*Ips sexdentatus*), again by various means. Alternatively, species such as the bark beetle *Pityokteines curvidens* may occur in forests that have been subject to some pollution stress but they are absent from areas with high to medium pollution levels. The situation is extremely complex and involves many interactions. As an example factors affecting the occurrence of outbreaks of the Pine bark bug (*Aradus cinnamomeus*) in Finland are shown in Figure 10[83]. Although some experimental work has been done (for example SO_2 treatment increases the susceptibility of Norway spruces to woolly aphid (*Adelges*) infestations[107]), much more work is required before the role of insects in forest decline can be fully determined.

The relationship between fertilisation by nitrogen and insect pests is fairly well established. Fertilisation results in increased rates of growth of insects and increased birth rates[164]. Low levels of nitrogen are

Figure 10. The role of air pollution in the development of *Aradus* outbreaks. (Source: Heliovaara, K. and Vaisanen, R. (1986). Industrial air pollution and the pine bark bug, *Aradus cinnamomeus* Panz. (Het., Aradidae). *Journal of Applied Entomology* **101**, 469-478).



matched by reduced levels of feeding by insects, together with slow rates of growth, long development times and poor survival[115]. Insect infestations may therefore occur in areas where there is increased deposition of nitrogen. Whether this is the result of the levels of nitrogen or the water content of the leaves/needles (the two are closely related[170]) remains unclear.

Multiple stress

This is probably the most popular hypothesis at the present time. No single mechanism is seen as responsible for the tree decline, rather the decline is the result of the cumulative effects of a number of stresses. These may vary between different areas[142]. There is a growing amount of evidence that some factors predispose a tree to damage whereas others incite or contribute to the damage[121]. 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 stresses. 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 possible that all three types of stress are involved in the decline of trees.

An example of a multiple stress hypothesis is provided by the 'explanation' of Norway spruce decline on acidic soils at high altitudes in West Germany[17,142,143]. It is proposed that during periods of high pressure in summer and early autumn, high concentrations of ozone build up. The ozone causes physical damage to the needle cells and, when rainfall occurs, nutrients such as magnesium, calcium, potassium and zinc are leached from the needles. The high acidity of the rainfall also results in increased leaching of calcium and magnesium from the soil. An inadequate supply of magnesium reduces the efficiency of chlorophyll synthesis in the needles, a process that is of crucial importance as a result of the intensive photo-oxidation of chlorophyll that occurs at high altitudes. The magnesium deficiency also results in lower rates of physiological activity in the needles and increases the frost susceptibility of older needles. Consequently, when frosts occur after long periods of comparative warmth, as occurred in 1979, 1981, 1982 and 1983, severe damage may occur and latent diseases in the trees may be activated. In north-east Bavaria, this process may be aggravated by the presence of high levels of sulphur dioxide. Further complications may occur as a result of the widespread occurrence of pathogens such as the needle-cast fungi mentioned previously. Combinations of stresses, such as that described above, may well be the principal cause of forest decline, a view that is receiving an increasing amount of backing from experimental work[16,113].

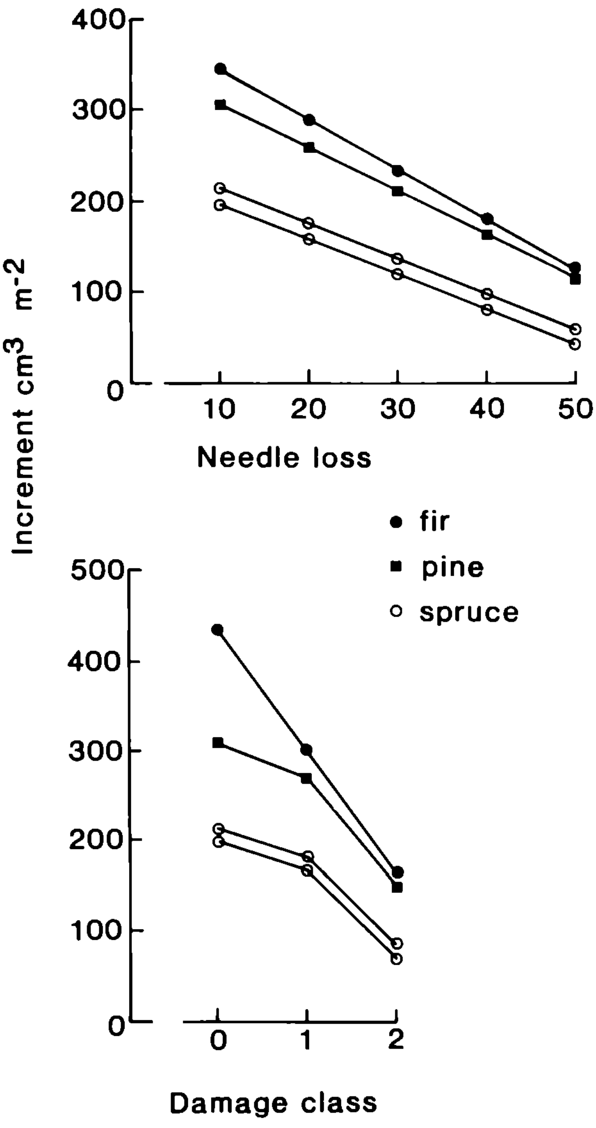
The Cost of Forest Damage

The economic impact of the forest decline is obviously of considerable interest but very little information is available. The effects can be divided into those occurring as a result of declining yield and those brought about by the release of large volumes of timber over a relatively short period of time. Both the owners of the standing timber and the wood-processing industry are likely to be affected. Further effects may occur on the infrastructures of regions where forestry is a major source of income, and additional problems may arise through the decline of tourism in badly affected areas and the decline of 'protection' forests.

Most attempts to establish the cost of the damage have used modelling. The value of the models is severely limited by the absence of suitable data. Little is known about the actual relationships between dieback, recovery and increment loss. However, information is now available on the relationship between needle loss and increment[49,73,112]. Some examples are shown in Figure 11. There appears to be a linear relationship between needle loss and increment reduction for needle losses of 10-50 per cent. However, for more severely damaged trees, there is now evidence that increment losses increase substantially when more than 50 per cent of the needles are lost[158]. This is consistent with physiological studies which indicate that one of the first responses of a tree under stress is a reduction in its annual increment. It must be emphasised that the relationship between crown density and increment remains controversial. This is because increment is affected by several factors, of which crown density is only one. Crown diameter and crown length must also be taken into account[9,162].

There is also little information on the effects of air pollution on wood quality. The available evidence is rather conflicting and probably depends on the exact nature of the material under test. Generally, the quality of the wood appears to be unaffected, although in some cases there may be slight reductions in mechanical

Figure 11. Increment reductions associated with crown needle loss (%) (From: Dong, P.H. and Kramer, H. (1986). Auswirkungen von Umweltbelastungen auf das Wuchsverhalten verschiedener Nadelbaumarten im nordwestdeutschen Küstenraum. *Der Forst-und Holzwirt* **41**, 286-290).



strength[169,184]. In addition, it seems that wood from damaged trees is drier than that from undamaged trees, and this may affect its storage properties. There is also evidence that damaged trees (particularly beech) are more susceptible to fungal attacks which may discolour the heartwood, thereby substantially reducing the value of the timber[57].

There is no indication that the increase in wood supply resulting from sanitation felling has significantly influenced market prices. This is because there has been a tendency to reduce the rate of felling of undamaged trees, which compensates for the unplanned felling brought about through sanitation work. However, there may be problems locally, particularly in areas where the damage is greatest. In such areas, the local market capacity may be saturated and it may not be economic to transport timber to distant markets. The alternative is to store the wood but, as already stated, wood from damaged trees tends to be drier than wood from undamaged trees. The use of techniques to postpone decay may then add significantly to the costs of storage.

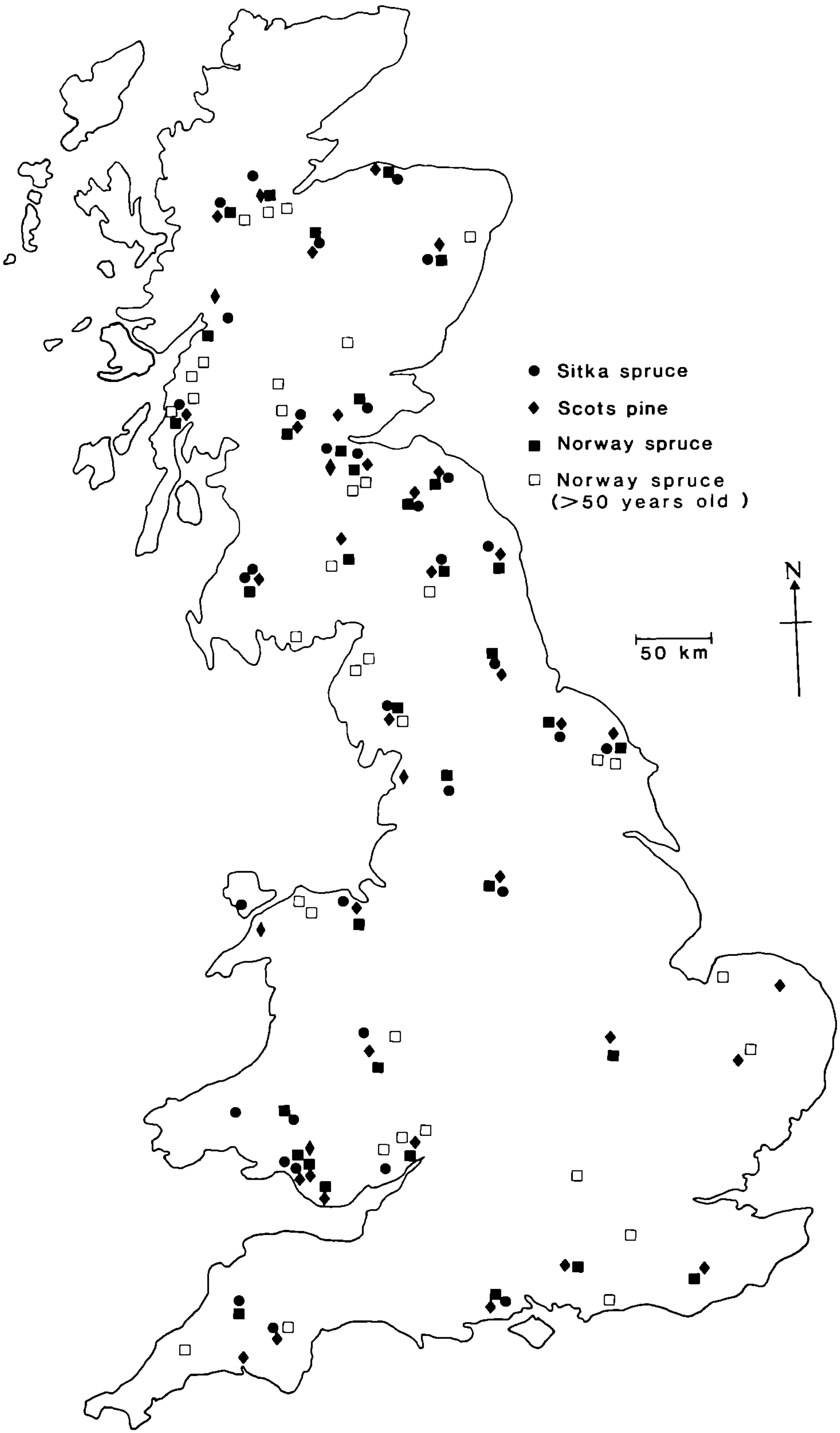
The impact of the decline of 'protection forests' is easier to assess. These forests occur predominantly in the high mountains and are maintained to prevent avalanches and reduce soil erosion. The economic impact of the decline can be assessed in terms of providing alternative protective measures. These include techniques such as terracing, barriers for retaining debris and structures designed to deflect avalanches, all of which are costly to erect and maintain.

Research by the Forestry Commission

The existence of forest decline on the European mainland has been established without doubt, although the causes remain unclear. The Forestry Commission has therefore started several major research programmes with the objectives of improving knowledge of the state of health of Britain's trees and investigating the effects of ambient levels of air pollution on the growth of important tree species.

The major tree survey has been directed towards conifers as damage was first recognised in these in central Europe. In 1986, it involved the survey of a total of 141 plots of Sitka spruce, Norway spruce and Scots pine throughout Britain (Figure 12). Norway spruce has been badly damaged on the continent and it is therefore useful as a link with surveys elsewhere. Sitka spruce has been included because of its importance as a

Figure 12. Approximate location of plots assessed in the 1986 forest health (air pollution) survey undertaken by the Forestry Commission.



commercial tree species in Britain and Scots pine because of its widespread distribution and importance in private plantations. In the 1984 and 1985 surveys, an upper age limit of 45 years was imposed as trees are commonly harvested within 50-60 years of planting in Britain. In 1986, the survey was extended to include older Norway spruce, up to 110 years old. This was done as there is evidence from Germany that it is the older trees that are affected first. The figures reported in 1986 are similar to those being reported from some continental countries.

In 1985, a study was begun with the aim of developing a method of assessing beech by recording the incidence of all the features by which this species might be expected to show ill-health. These include various symptoms which have been associated with the decline of beech in West Germany: leaf yellowing, premature leaf fall, leaf shape and leaf rolling, crown thinness, abnormal branching patterns and insect and fungal damage. Early leaf yellowing, crown thinness and abnormal branching were observed quite frequently. None of the observed damage was considered to be outside the range expected for beech in the UK, particularly in view of the effects of the series of droughts in 1975, 1976, 1983 and 1984[117]. However, damage by pollution was not ruled out as a possible cause of some of the symptoms and further studies are underway.

In 1986, an investigation was begun into the health of hedgerow ash and oak. No results are yet available.

The effects of pollutants on trees are being investigated in an experimental project started in 1984. Open-top chambers are being used to grow trees in filtered air and in ambient air. They also allow the response of trees to precisely determined levels of pollution to be investigated. The study has a number of aims, including establishing whether trees are being affected by ambient levels of pollution in various parts of the UK and the long- and short-term concentrations of pollutants required to induce damage in trees. The main species involved in the trials are Sitka spruce, Norway spruce, Scots pine, European beech and oak, although other species are being examined where appropriate. This study is likely to be particularly valuable because of the care with which plant material under test has been selected and as a result of the large number of chambers involved (16 at each of three sites).

Conclusions

Forests are subject to a wide variety of stresses of which air pollution is only one. Evidence from the European mainland strongly suggests that air pollution is involved in the decline of forests that has been observed there over the last few years. In Britain, detailed monitoring projects are under way to ensure that if a decline similar to that on the continent does occur, it will be identified in its early stages. Acidification has been occurring at an accelerated rate in areas such as the Pennines since the Industrial Revolution. The introduction of the tall-stack policy following the Clean Air Acts of 1954 and 1962 has led to marked decreases in pollution in urban areas but may have contributed to pollution at remote sites in Britain and in Scandinavia. The presence of trees in these areas may enhance the deposition rates of acidity, leading to a decline in water quality. Acidification of freshwaters has certainly occurred in several places, although the role of afforestation in this process is far from clear.

The forest decline on the continent has continued throughout the early 1980s, and is now affecting both conifers and broadleaves. The area affected has also increased. The causes of the decline are still unknown, but one of the most likely explanations is a multiple stress hypothesis involving air pollution as an inciting or predisposing stress, adverse soil conditions as a predisposing stress and short-term climatic factors as inciting stresses. Although adverse weather conditions may have triggered the decline, it seems probable that this decline would not have occurred if the trees had not already been weakened by air pollution or some other stress.

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