FORESTRY COMMISSION
OCCASIONAL PAPER 42

Forestry Expansion – a study of technical, economic and ecological factors

The Impacts of Forestry Expansion on Water Quality and Quantity

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INTRODUCTION

The UK water resource includes almost 1500 discrete river systems which comprise approximately 200 000 km of water courses with some 5000 lakes and reservoirs plus a variety of aquifers ranging in size from small areas of river gravels to the chalk. This water resource is utilised and exploited for a wide range of purposes: it forms the source of water for human consumption, for use in industry, as a coolant in power generation, in recreation (boating, swimming and fishing) and is the habitat for a wide variety of aquatic organisms. Changes in the quantity and quality of surface or ground waters can influence any or all of the above functions; forestry, as a major land use influences both water yield and quality.

Water supply for houshold and industrial use

In 1981 some 32 100 million litres of water were extracted per day; approximately 50% for public water supply, 17% direct industrial abstraction and 33% CEGB cooling water. Considered as a whole the UK receives more than enough precipitation to satisfy this level of demand. However, there is a mismatch between the distribution and timing of precipitation inputs and the location and level of demand. The bulk of the precipitation falls in the uplands of the north and west while the main demand is in the populous lowland areas of the south and east. Table 1 shows the variation in precipitation input to a number of regions of the UK, evaporation losses, runoff and use in water supply; in Scotland only 1% of the available 'runoff resource' is used compared with 20% in south-east England. This mismatch is largely overcome by the storage and distribution systems developed by the water industry.

Table 1 Water balance for parts of Britain ($m^3 \times 10^8$).

	Rainfall	*Evaporation*	*Runoff (R)*	*Supply (O)*	$\frac{O}{R}$ %
Scotland	1080	295	785	8.7	1.4
N. Ireland	149	56	93	2.5	2.7
Wales	272	99	173	4.2	2.4
W. England	903	385	518	28.3	5.4
SE England	442	289	153	30.8	20.1

Water supplies are taken from springs, lakes, reservoirs, streams, rivers and groundwater sources. Direct supply from reservoirs and river abstraction are by far the dominant sources on a national basis. However, in some areas of the UK, groundwater forms the dominant source (Figure 1.) The variation in the proportion of supplies obtained from groundwater reflects the locations of the major aquifers, the largest are the chalk of the south and east of England and the Bunter and Keuper sandstones of the midlands and north-west but important sources are also found in shallow aquifers such as river gravels. The UK uplands are almost devoid of any major underground water sources. It is interesting to note that the major aquifers mainly underlie areas of intensive agricultural use with relatively little woodland cover at present. The relative contribution of the various surface water sources also varies regionally. The major conurbations of south Wales, Birmingham, Liverpool, Manchester and south Yorkshire draw much of their water from direct supply reservoirs in the uplands of Wales, the Pennines and the Lake District. Similarly, direct supply from reservoirs and lochs is the dominant source of supply in Scotland, accounting for some 89% of developed sources (Scottish Development Department, 1985). Water from these direct supply reservoirs is fed by gravity via pipes to treatment works close to the cities and towns where the water is used; very little treatment is generally required (filtration and disinfection) and as a result this forms the cheapest source of potable water, costing approximately £5 per megalitre compared with £20-£50 per megalitre for groundwater or river abstraction. In total some 1% of the UK land surface drains into direct supply reservoirs. In the south and east of England water supplies are mainly obtained from a combination of river abstraction and groundwaters. The amount of water which can be abstracted at a given point is influenced by the need to maintain adequate flow downstream to dilute inputs, e.g. from sewage works, and to support the biological regime of the river and navigation. Many of the major rivers of the UK rise in the uplands of the north and west and reach the sea after flowing through the densely populated lowlands with their large concentrations of industry and intensive agriculture. Any significant reduction in the flow and/or water quality of these upland waters reduces their value in flushing and diluting the pollutant inputs downstream.

Figure 1 Percentage of water supplies obtained from ground water sources (Water Authorities Association 1988 and Kirby 1984)

National overviews of water resources and sources can overshadow the role, and importance of a given source to a particular community or individual. In many areas, the water industry can find alternative supplies if one source has to be taken out of use because of quality or quantity considerations. However, in remote rural areas, e.g. in upland Scotland or the islands there may be no alternative if a supply is polluted or its flow drastically reduced. Considerations of the impacts of land use on the water resource need to be sensitive to local as well as national needs and consequences.

Demand for water in the UK is said to be growing at *c.* 1% per annum (Kirby, 1984). However, most of the increase is in England and Wales as a review of public water supplies in Scotland (Scottish Development Department, 1985) suggests that demand there may fall over the next 30 years; the review also suggests that 'for some considerable time to come, developed resources are adequate to meet demands at both national (Scotland) and regional levels.' Current demand in Scotland accounts for *c.* 70% of developed resources. Increased demand in England could be met by further abstraction from rivers or aquifers, from direct supply reservoirs, or by the introduction of 'new' technology, e.g. desalination. River abstraction in parts of the south and east of England cannot be increased much more. For example, it is predicted that, by 2001, natural flow in the Thames in dry periods will be too low to satisfy demand. The deficit could be met by supply from surface storage reservoirs, transfers from other rivers or increased groundwater abstraction. The scope for increased groundwater abstraction is limited, especially in the Thames basin but could be increased given further development of groundwater recharge, as now carried out in the Lea Valley Scheme. Transfer from other river systems is being increasingly considered; the ultimate development would be a national water grid. This would enable existing resources in the wetter north and west to be used to maintain flow in the rivers of the drier south and east, thus permitting increased river abstraction in these latter areas. Table 1 suggests that there is still a large untapped water resource in Scotland and Wales. In the longer term, additional storage reservoirs may be needed but their siting will depend on whether or not a national water grid is developed. Construction of further large reservoirs is also likely to be opposed by environmental and conservation pressure groups.

Hydroelectricity generation

Hydroelectricity accounts for some 2.2% of total electricity generation on a UK scale (1.0% from pumped-storage schemes and 1.2% from other hydroelectric plant) but is important in parts of Scotland with six stations with generating capacities over 45 MW operated by the North of Scotland Hydro-Electric Board.

Generation of hydroelectricity is clearly dependent on a plentiful supply of water. Such generation plants are carefully sited to ensure suitable conditions but catchment management following construction of the generation plant may influence water yield and flow regimes with consequent adverse impacts on power output. Barrow *et al.* (1986) state that most of the prime sites for hydroelectric power in the UK have already been developed and that future developments may be of the 'runoff-river' type. Existing plants are mainly located in the uplands of Scotland and Wales.

Recreational use of surface waters

It is extremely difficult to evaluate the recreational and habitat uses/functions of waters in the UK. Angling is said to be the largest participator sport in Britain and its value in terms of fishing licences, rental of fishing rights, supply and manufacture of fishing tackle and tourism linked to angling, amounts to hundreds of millions of pounds annually. The industry is clearly dependent on the existence of waters which provide a suitable habitat for fish; the requirements vary between fish species but include flow, and water quality considerations plus the existence, in self sustaining fisheries, of suitable spawning grounds. A study of the economic value of sporting salmon fishing was carried out for private beats in the Kyle of Sutherland, Tay and Spey salmon fisheries districts in 1984: this calculated the annual expenditure by fishing parties on permits, accommodation etc as £0.66m, £4.27m and £7.16m for the three districts respectively. (Tourism and Recreational Research Unit, 1984). (Radford, 1984), using a discount rate of 10% and a time horizon of 10 years, calculated the total value of salmon fisheries on the Wye, Mawddach, Tamor and Lune as £28.7m, £4.9m, £15.8m and £2.3m respectively, at 1984 prices. (The value of fisheries is considered further in McGilvray and Perman, 1991). These studies show that the financial impacts of a loss of game fisheries would be considerable. Most of the high value brown trout, sea trout and salmon fisheries are located in upland areas in waters with low solute and pollutant loads while the major coarse (cyprinid) fishing waters are in the lowlands where the waters have a relatively high load of dissolved solutes. The main salmonid and cyprinid rivers in the UK have been designated in response to an EEC directive issued in 1978; 74% of the designated 45 800 km of salmonid waters are in Scotland, 23% in England and Wales; 96% of the designated 5 500 km of cyprinid waters are in England and Wales.

The use of waters for swimming and boating requires waters which do not present a health risk and which do not have serious aesthetic constraints, e.g. smell. The impact of changes in water quality on these forms of recreational use was demonstrated during the summer of 1989 when boating and other recreational uses were banned temporarily on a number of eutrophic water bodies in central and eastern England because of potential risks to health associated with algal blooms triggered by increased PO_4 and NO_3 levels and high temperatures.

Water quality

Water quality can be interpreted in a variety of ways; the broadest interpretation includes any aspect of water which influences its use by man or its functioning as a natural habitat for aquatic biota. This broad definition would include chemistry, dissolved oxygen, smell, taste, colour, sediment and temperature. Water chemistry itself covers many solutes but those of particular concern are nitrate, ammonium, phosphate, aluminium, lead, manganese and organic compounds (natural and man made). The relative importance of the various elements of water quality will vary with respect to different uses or functions of the water. For example, dissolved oxygen, acidity, calcium and aluminium are important controls on invertebrate and fish populations, lead, aluminium, nitrate and ammonium are important from the point of view of human health: taste and smell are important aesthetic considerations in drinking water supplies although they may not be linked to any health risk. Water used in boilers at pressures up to 35 atmospheres must be particularly high purity and with solutes and alkalinity within a specific range; the textile industries need 'soft' water while the brewing and paper industry need 'hard' water. In the past these industries were sited in areas with naturally suitable water but today the water chemistry is adjusted as and when necessary.

Figure 2 Location of sites discussed in this paper

Surveys of river quality in the UK are usually related to generalized water use objectives. Assessment is based on chemical criteria and qualitative comments on contents of toxic substances, but the classes are often quoted in terms of the biological oxygen demand (e.g. Table 2), and based on proposals by the Royal Commission on Environmental Pollution. In Scotland, rivers have also been 'scored' biologically since 1968. In 1987, 65% of rivers and freshwater canals in England and Wales were classified as of good quality, 24% as of fair quality, 9% poor and 2% bad quality (Department of Environment, 1989). In 1980, 94.6% of Scottish rivers were classified as unpolluted (≡ (good quality). These surveys and chemical criteria, mentioned above are designed to produce general overviews of the state of the Nation's rivers and to pinpoint heavily polluted waters.

Table 2 Classification of rivers – as defined by the Department of the Environment (DoE).

Class 1	Unpolluted or recovered from pollution. Biological oxygen demand (BOD) generally less than 3 mg l^{-1} and well oxygenated.
Class 2	Doubtful quality. Reduced oxygen content, containing turbid or toxic discharges but not seriously affected.
Class 3	Poor quality. Dissolved oxygen below 50% saturation for considerable periods, occasionally toxic and changed in character by the discharge of solids.
Class 4	Grossly polluted. BOD greater than 12, incapable of supporting fish life, smelling and offensive in appearance.

National overviews of river chemistry are also available from the Harmonised Monitoring Scheme which involves 250 sites in GB and regional surveys of 'water quality' are also available for parts of the UK. Summary maps of the distribution of acid waters and acid sensitive waters are also available (e.g. United Kingdom Acid Waters Review Group, 1989).

Particular chemical criteria, and limits apply with respect to specific uses/functions of waters, e.g. for human consumption or as fisheries. UK legislation covering the control of discharges into waters and criteria for water for human consumption are contained in the 1989 Water Act, associated regulations and DoE *Guidance Notes.* An EEC directive on drinking water *(Directive 80/778/EEC)* was issued in 1980 and contains guide levels and maximum admissible concentrations (MAC) for 62 parameters. The MAC values for selected parameters are given in Table 3. The MACs have now been largely included in the Water Act, as 'prescribed concentrations or values' of properties, elements and substances which affect the wholesomeness of surface waters. National governments within the EEC have been able to grant derogations to waters undertakings to exceed MACs for limited periods but these standards will be mandatory in the next few years. The draft guidance on safeguarding the quality of public water supplies issued by DoE subsequent to, but linked to the Water Act, provided for the designation of nitrate sensitive areas and, hence the MAFF nitrate sensitive scheme.

Table 3 Guide and maximum admissible levels for selected parameters in the EEC drinking water directive.

	Guide level	*Maximum admissible level*	*Units*
A. Organoleptic parameters*			
Colour	1	20	mg l^{-1} Pt/Co
Turbidity	1	10	mg l^{-1} SiO_2
	0.4	4	Jackson units
B. Physico-chemical parameters			
Chlorides	25	-	mg Cl l^{-1}
Sulphates	25	250	mg SO_4 l^{-1}
Aluminium	0.05	0.2	mg Al l^{-1}
C. Substances undesirable in excessive amounts			
Nitrates	25	50	mg NO_3 l^{-1}
Nitrite		0.1	mg NO_2 l^{-1}
Ammonium	0.05	0.5	mg NH_4 l^{-1}
Phenols		0.5	μ Mn l^{-1}
Manganese	20	50	μ Mn l^{-1}
D. Toxic substances			
Arsenic		50	μ As l^{-1}
Cadmium		5	μ Cd l^{-1}
Nickel		50	μ Ni l^{-1}
		–	
E. Microbiological			
Total coliforms			MPN < 1 (multiple tube method, sample volume 100 ml)

*Organoleptic – a stimulus capable of affecting the sensory organs

The 1978 EEC Directive on salmonid and cyprinid waters now includes limit and/or guideline values for 13 parameters (Table 4). In 1985 approximately 2% of the designated salmonid waters did not comply with EEC criteria and 9% of the cyprinid waters. Maps of the designated waters are available for each water authority in England and Wales; Scotland is covered by four maps and Northern Ireland by one.

Table 4 Parameters for which limit and/or guid values are contained in the EEC directive on salmonid and cyprinid waters.

Temperature	Dissolved oxygen	Suspended solids
BOD	Phosphorus	Ammonia (non-ionised)
Ammonia (total)	Phenols	Petroleum hydrocarbons
Zinc	Nitrates	Dissolved chlorine
pH		

The water quality requirements for particular aquatic biota are now being established and can be seen as water quality criteria for the maintenance of a given habitat. It is surprising that the parameters listed in Table 4 do not include aluminium although it is now recognized that concentrations of > 0.1 mg l^{-1} are toxic to salmonids in soft waters.

The relative importance of the various aspects of water quality, and those which give rise to the greatest concerns in the water industry will vary with time, between water authorities and even between divisions in a given authority. Colour is currently a particular problem to the Yorkshire and Severn-Trent Water Authorities; nitrates are of main concern to the East Anglian and Severn-Trent Authorities; acidity and aluminium are of greatest concern to the environmental and fisheries divisions of the water undertakings in Wales and parts of Scotland; phosphate and suspended sediments are of concern in the Welsh and Severn-Trent areas and with Scottish water undertakings.

Organic materials in waters can be of concern for a variety of reasons. Dissolved humic materials give rise to colour which is generally unacceptable in drinking water. Suspended organic materials raise similar problems to other suspended sediments. Solvent and pesticide residues, soils and petroleum products can give rise to problems of taste and smell but also, in some cases, problems related to human health and toxicity to aquatic biota. Organic materials from sewage and agricultural wastes can result in deoxygenation with impacts on aquatic biota.

Acidity and aluminium have become matters of concern in the last 10 years, the colour problem has increased since about 1976, nitrates over the last 20 years, solvents and pesticide residues in groundwaters have been recognised over the last 5 years. These changes can reflect changes in technology or land use, identification and definition of health or fish toxicity problems, or developments in analytical technology.

The greatest current concerns in upland surface waters are probably colour, sediment, aluminium, acidity and phosphate; in lowland surface waters, nitrate, phosphate, sewage and farm waste effluents; in groundwaters, nitrate, organic solvents and pesticides residues.

The levels at which a given component of water quality gives cause for concern in the water industry will vary with current legislation on water quality use or function, treatment requirements and costs. For example, as noted earlier, waters from direct supply reservoirs generally require very little treatment and may only comprise of filtration and disinfection, and therefore treatment costs are low. A small increase in colour, sediment, or algae in these waters may result in major increases in treatment costs. Increases in colour in Upper Nidderdale, North Yorkshire between 1979 and 1983, resulted in additional treatment costs for Yorkshire Water Authority of £34,000 per year at 1983/84 prices (McDonald and Kay, 1987). The Authority has embarked on a £30 million investment programme in treatment plants to reduce colour to acceptable levels in all its supplies. River abstraction plants may have more sophisticated treatment works and higher treatment costs are anticipated from the outset.

IMPACTS OF FOREST MANAGEMENT AND TREE GROWTH

Evaporation losses

Research on the impact of forests on water yield and streamflow has now been in progress for over 40 years. In the 1950s Law (1956, 1957), working in the catchment of Stocks reservoir, northern England, showed that evaporation from forested land was greater than from grassland. Based on the results of this work, it was suggested that the forestry enterprise would need to pay the water undertaking a rent of £500 ha^{-1} yr^{-1} to cover the loss of water. Walsh (1980) developed the assessment further and calculated that, if 80% of the reservoir catchment was afforested, the yield of the reservoir would be reduced from 85 to 59 megalitres at a cost to the water industry of £170,000 to cover provision of water from other, more expensive sources.

The Plynlimon catchment study (Newson, 1979) compared water yields from adjacent grassland and forested catchments (60% forest cover). Evaporation from the grassland catchment was equivalent to about 17% of incoming precipitation. From the forested catchment evaporation accounted for 38%, resulting in a reduction in streamflows of approximately 15%. High evaporation losses from forests have also been shown in small catchment studies in Kershope Forest (Cumbria), 50% of incoming rainfall (Anderson *et al.*, 1990) and Beddgelert Forest (north Wales), 32-40% (P.A. Stevens, personal communication). The work at Kershope has also shown that streamflow increases rapidly again by (290 mm yr^{-1}) following clear felling.

Process studies have shown that the major difference in evaporation between forest and non-forest vegetation is due to interception and subsequent evaporation of the incoming rainfall by the forest canopy (Calder, 1976). Indeed, Roberts (1983) suggests that transpiration losses from temperate forests in Europe are relatively constant at about 300-350 mm. Interception loss increases with rainfall, windiness and canopy density. Losses of 10-60% of incoming rainfall, on an annual basis, have been measured in European studies. In general losses from deciduous species are less than from evergreen (Institute of Hydrology, 1989) but there is a large variation between species. At Gisburn Forest, losses from Scots pine were significantly greater than those from Norway spruce, oak or alder (Cape and Brown 1986). (Brechtel, 1976) has shown the impact of variations in canopy 'cover' on interception where incoming rainfall amounted to 3.6 mm per day; losses increased from 1-2% to 32-39%, as stand basal area increased from 5 to 19-22 m^2/ha.

The results from some of the earlier studies have been questioned by data from the Balquhidder catchment studies in central Scotland. At this site, a moorland with 66% heather and 34% grass cover showed evaporation losses of 24% while a nearby catchment, with 40% forest cover and 60% grassland, showed losses of 19%. Canopy interception studies at Balquhidder have found losses similar to those reported for other upland forests in the United Kingdom (Johnson, 1990) so other factors may explain the contrasting results from Plynlimon and Balquhidder: (1) heather, important at Balquihidder but rare at Plynlimon, shows evaporative losses closer to those of trees than grassland (Wallace *et al.*, 1982; Hall, 1987), (2) the grassland in the upper parts of 'forest' catchment at Balquihidder has a shorter growing season, due to severe climate, and hence lower evaporation losses than the grassland at Plynlimon.

A number of models have been developed for the investigation of evaporation from wet heather, grass, and forest and in snow conditions, at the annual, daily and seasonal scale (Calder, 1990). (Calder and Newson, 1979 and 1980) and (Calder, 1985) have developed simple models for practical applications on an annual basis, and which can be used to predict interception loss following afforestation of part or all of a catchment. This model is based on an interception loss from forests of approximately 35% of annual rainfall: the model predicts that, in the wet upland regions of the UK, afforestation of 75% of a catchment will result in reduction in runoff of 20%.

Although the existing models provide broad indications of likely changes in water yield following afforestation in the wet uplands, refinements are necessary to allow for (1) species differences, both of trees planted and of pre-planting vegetation, (2) variations in the proportion of precipitation falling as snow. The practical models will also require modification for application in the drier uplands.

The predicted reductions in stream flow following afforestation in the uplands should be viewed over the period of a complete rotation. Thus, the Coalburn study has shown pre-afforestation ploughing and drainage to result in a net increase in water yield of 5% over the first 10 years. (Calder, 1990) has taken into account variations in forest cover and age over the period 1960 to 2025 in a study of the implications of planting, on the Crinan Canal, west Scotland using the simple (Calder and Newson , 1979) model. Planting was begun in 1960 and runoff is predicted to decrease gradually from 1970 to give a reduction of 26% by the year 2000, remaining at this level until 2020, before increasing again to 16% in 2025.

The impact of increased evaporation as a consequence of afforestation on the water resources of the dryer lowlands needs further assessment. Although the proportion of the incoming rainfall lost may be smaller than in the uplands it may have a major impact on aquifer recharge and runoff. Current studies by the Institute of Hydrology, supported by DoE, on the hydrological impacts of hardwood plantations in lowland England will provide valuable data.

Atmospheric deposition

The deposition of gases, particulate materials, fog, mist and cloudwater from the atmosphere are influenced by the nature of the vegetation surface. Where forests replace low growing vegetation there can be a large net increase in the inputs of both pollutants and non-pollutants. Deposition of gases ($S0_2$, $N0_2$, N0, O_3, HNO_3, HCl and NH_3) to forest canopies is influenced by a number of processes and the dominant process varies between gases (Fowler *et al.,* 1989). NH_3, HNO_3 and HCl are readily absorbed by leaf and stem surfaces and their deposition is very sensitive to surface roughness and therefore strongly influenced by replacement of low, smooth canopies by forest. Sulphur dioxide and O_3 are absorbed through stomata and changes in vegetation canopy will not have major impacts on deposition. However, recent work in the Netherlands has investigated co-deposition of NH_3 and SO_2 to leaf surfaces. Where ammonia concentrations are significant this may be a more important pathway for SO_2 deposition than absorption through stomata and it will be sensitive to surface roughness, through the effect of this on ammonia deposition. Fluxes of NO_2 have been measured to and from canopies while the NO flux is usually from soils; there seems insufficient data on which to draw conclusions about the impact of changes in vegetation on the deposition of these two gases.

The influence of canopy structure on the deposition of particulates, mist, fog and cloudwater varies with droplet size. Cloudwater droplets in the range of 5-40 μm are efficiently collected by foliage via impaction and sedimentation, particularly in wind speeds of 5-10 m s^{-1} ; this can be an important pathway for deposition in areas of western and northern Britain where driven cloud frequently envelops the higher forests. (Figure 3). The concentrations of solutes in clouds and mist droplets are also considerably higher than in rainfall. The deposition to forest canopies will, therefore, be influenced by local atmospheric chemistry and the frequency of low cloud and fog.

As a result of the above interactions the planting of trees on grassland will lead to increased dry deposition of NH_3, HNO_3 and HCl; the NH_3 could affect SO_2 inputs. These effects will be greatest in the lowlands where concentrations of these gases are highest. An extreme example can be taken from forest sites in the Netherlands situated downwind of intensive livestock enterprises; NH_4 and $N0_3$ inputs at one such site were double those on adjacent bare ground with total deposition to the forest of 100 kg ha^{-1} yr^{-1} (Mulder, *et al.,* 1987). Forestry replacing moorland, in those upland areas with frequent low cloud, mist or fog, will lead to increased occult and aerosol deposition. This will primarily affect inputs of sulphate, nitrate and sea salts.

Quantification of the effect of tree planting, and of the variation in deposition to different species (see below), has been attempted by (1) comparing element fluxes below the canopy with bulk precipitation inputs (2) catchment based budget studies and (3) modelling approaches. Each of these approaches has its drawbacks and problems. For example, rainfall chemistry is influenced by a number of processes as it passes through the tree canopy, thus a wide range of solutes can be leached from the canopy in addition to wash-off of dry and occult deposition thus confounding calculation of 'excess' deposition using this approach. However, recent studies using radioisotopic tracers suggest that only *c.* 5% of SO_4 in throughfall is leached from the canopy the rest being derived from atmospheric deposition of S to the canopy (Lindberg and Garten, 1988; N. Cape, personal communication).

Comparative catchment studies at Plynlimon show net sulphate and chloride export from upland grassland to be 21 and 30 kg ha^{-1} yr^{-1} respectively but 36 and 44 kg ha^{-1} yr^{-1} from a nearby catchment with 48% forest. This suggests a 30-50% increase in fluxes due to additional deposition on to the forested part of the catchment. Removal of the forest

Figure 3 Distribution of occult deposited sulphur (mg m^{-2})
(D. Fowler personal communication)

canopy at felling leads to a reduction in output; at Kershope net sulphate-S and chloride outputs in the first year after felling were 39 and 60.9 kg^{-1} ha^{-1} yr^{-1} respectively from felled plots compared with 53 and 118 kg ha^{-1} yr respectively from adjacent unfelled controls (Adamson *et al.*,1987). Similar results have been reported from Scandinavia (H. Hultberg, personal communication). However, long term catchment studies, at Loch Dee and Llanbrymair, commencing soon after planting, have not shown any increase in fluxes since planting.

Fowler *et al.*, (1989) have used a modelling approach to predict the effects of afforestation on inputs of S and N from the atmosphere to Kielder. Their calculations suggest that at an altitude of 300 m and with 1000 hr of low cloud per year, S inputs would increase from 17.4 to 22.6 kg ha^{-1} yr^{-1} and N inputs from 12.4 to 23.4 kg ha^{-1} yr^{-1}. The increase in S deposition is mainly accounted for by greater interception of cloudwater and the increase in N by both greater interception of cloudwater and dry deposition of gases. A similar approach could be used to predict the effects of afforestation at any upland site in the UK given the required input data.

Atmospheric deposition to forest canopies is also affected by tree species. For example, excess sulphate-S deposition under oak, alder, pine and spruce at Gisburn was 2.6, 4.5, 11.5 and 30.0 kg ha^{-1} yr^{-1} respectively; 'excess' deposition being calculated as (throughfall + stemflow fluxes) – bulk precipitation inputs (Cape and Brown, 1986). In general, occult deposition and dry deposition of NH_3, HNO_3 and HC1, and SO_2 if co-deposited with ammonia, is greater to coniferous canopies than to deciduous hardwoods at a given site (e.g. Mayer and Ulrich, 1978; Nys *et al.*, 1990); thus, the inputs of sulphur to beech forests at Sollingen (West Germany) were 45-51 kg ha^{-1} yr^{-1} compared with 80-86 kg ha^{-1} yr^{-1} to adjacent spruce. There is, however, considerable variation between hardwood species and between conifers (Cape and Lightowlers, 1988). Deposition of S and sea salts to larch is greater than to pine or spruce (Reynolds *et al.*, 1989).

Primarily as a result of the above mechanisms, concentrations and fluxes of Na, Mg, NO_3, SO_4 and C1 are greater in drainage waters from afforested catchments than from non-planted catchments on similar soils and bedrock. However, these increases themselves are not sufficient to produce water quality problems; it is the impact on soil-water interactions of these increased solute inputs to acid soil systems and consequent changes in drainage water acidity and aluminium which are of concern.

Acidification

In recent years there has been intense debate on the possible impacts of afforestation on the acidity and/or aluminium concentrations of surface waters. Results from a number of paired catchment studies and from stream surveys in Wales (Ormerod *et al.*, 1989) suggest that streams draining forest are more acid and/or have higher concentrations of aluminium than streams draining moorland or grassland on similar soils and geology. The differences in acidity and/or aluminium level are most pronounced in high flows. Palaeolimnological studies in sensitive areas with significant pollutant inputs also suggest an increase in the rate of acidification following afforestation (Kreiser *et al.*, 1990). The effect is only seen where acid, base poor soils overlie massive rocks containing few readily weathered minerals, such as silicates or carbonates, in areas with significant levels of atmospheric pollution. Thus, forestry is not seen as the primary cause, rather atmospheric pollution. Such conditions occur in large areas of the uplands of Wales, parts of Scotland, the uplands of south-west England and the Lake District (Figure 4).

Figure 4 Distribution of surface waters susceptible to acidification (stippled). Information for Scotland is derived from maps in Wilson et al. (1989), for Wales from Hornung et al. (1990) and for England from Kinniburgh and Edmunds (1984) and the Soil Survey and Land Research Centre

The reported increases in acidity and/or aluminium have been linked to changes in streamwater biota; particularly to a reduction, or disappearance, of salmonid fish populations (e.g. Stoner and Gee, 1985; Harriman and Morrison, 1982) and a reduction in diversity of invertebrate populations (Ormerod and Wade, 1990). These trends can be seen as an intensification of changes which had already taken place as a result of the impact of acidic deposition in these sensitive areas. Egglishaw *et al.*, (1986) have also presented data which they suggest show a link between recent declines in catches of adult salmon in some parts of Scotland and the amount of forestry in the nursery streams, although the authors were unable to link the decline to any specific aspect of water quality.

There are a number of problems inherent in comparative catchment studies (Nisbet 1990). No two catchments have identical relief, soils, geology and hydrology and it is difficult to separate the impact of variations in these factors on drainage water chemistry, from any due to afforestation. (Nisbet, 1990) also stresses that results from a number of long term studies established soon after, or before planting; have not shown increases in acidity or aluminium concentrations.

A number of mechanisms could produce the reported increase in acidity and/or aluminium concentrations: (1) the increased deposition of acidic pollutants following development of the forest canopy, (2) changed soil hydrology and water pathways, as a result of ploughing and drainage and soil drying, (3) increases in solute concentrations as a result of greater evaporation from the forest, (4) production of mobile anions due to increased mineralization of organic S and N, resulting from drying, (5) soil acidification due to base cation uptake by trees, (6) the production of organic acids from decomposing needles. The dominant mechanism would seem to be increased inputs of acidic pollutants leading to mobilisation of protons and/or aluminium from the acidic soils but further research is needed to quantify the role of each mechanism.

The sensitive sites on which planting might cause an increase in acidity and/or aluminium concentrations of drainage waters can be identified from information on soils and geology, (Hornung *et al.*, 1990). The Welsh Water Authority have suggested that planting should be limited in such sensitive areas (Table 5). An alternative strategy is to lime part of the affected catchment to ameliorate any additional effects of planting. A number of UK research projects are currently examining liming methods and their environmental impacts.

Table 5 Suggested guidelines for acceptable conifer afforestation in upland Wales (J. Stoner, personal communication)

Mean hardiness (mg l^{-1} Ca CO_3)	*Acceptable conifer afforestation*
12	No planting other than small catchments drained by first order streams having a negligible effect on receiving water course.
12-15	Up to 30% of catchment, subject to current Forestry Commission guidelines.
15-25	Up to 70% of catchment, subject to current Forestry Commission guidelines
25	Observation of current Forestry Commission guidelines.

Ground treatment Site preparation for afforestation on many upland soils necessitates ploughing and drainage of the ground. The turf ribbon removed from the plough furrow is inverted and the young trees are planted into it, the ribbon being a more suitable environment for nutrient release and root growth than undisturbed soil. The ribbon also benefits early tree growth by suppressing competition from ground vegetation. In recent years the practice has been to create furrows which run at right angles to the contour and to intercept these with collecting drains at sufficient frequency to prevent furrow run-off building up to excessive levels. In addition cut-off drains may be required to intercept water flowing on to the area from springs or adjacent ground. The drains which, following heavy rain do carry large quantities of run-off, are aligned almost level with the contour to reduce run-off velocity and therefore soil erosion.

It is inevitable that the exposure of bare soil in the process of site preparation promotes erosion. Forestry drains and plough furrows are unlined channels which may penetrate through the surface soil horizons to unconsolidated material below and moving water may scour material from their base. Unlike lowland agricultural drains they are open and therefore material may fall from their sides (for example as a result of frost action) and be transported away in drainage water.

Erosion and sediment yield in forests have been the subjects of two recent reviews (Moffat, 1988; Soutar, 1989) and both these authors state that soil erosion can have serious consequences for water resources, although any marked increase in stream sediment load could be detrimental to the conservation value of streams. Three studies are quoted where suspended sediments were measured before and after ploughing. At Llanbrynmair in mid Wales suspended sediment increased from 37 to 90 kg ha^{-1} yr^{-1} in the first 14 months after ploughing in one sub-catchment and from 7 to 31 kg ha-1 yr^{-1} in another in the first 6 months after ploughing. At Coalburn in northern England suspended sediment increased from 30 kg ha^{-1} yr^{-1} before ploughing to 240 kg ha^{-1} yr^{-1} over the first 5 years after ploughing and then fell to 120 kg $ha^{-1}yr^{-1}$ in the years immediately following. At Holmstyes in northern England suspended sediment increased from 32 to 513 kg ha^{-1} yr^{-1} in the first year after ploughing.

Site preparation has the potential for affecting water quality in other ways. For example, at Nant y Moch in mid Wales a stream draining a catchment which had recently been ploughed had elevated concentrations of sulphate, nitrate, ammonium and aluminium compared with a nearby unafforested stream. Colour was greater in the stream on the ploughed catchment and the pH of the stream on the ploughed catchment averaged 4.2 compared with 5.6 on the unafforested catchment. These differences have been attributed to aeration of the soil leading to the oxidation of sulphur and nitrogen compounds which changed the ion exchange equilibrium causing various ions to be leached from the soil. (J. Underwood, personal communication). At Llanbrynmair marked differences in stream solute concentrations were not found between ploughed and unploughed catchments except in the cases of potassium and silicon which were lower on the ploughed catchment.

At the time of planting some road construction usually takes place although the complete road network may not be installed until the thinning or clear felling stages are reached. Construction, particularly in steeply sloping terrain, can require large scale excavations with the excavated material being used to construct embankments. Exposed surfaces may be slow to revegetate and this, coupled with the erosive potential of the water which gathers on the surface of the road and of water courses over which roads must pass, has in the past led to quality problems in streams and rivers. However, the impact of road construction is often difficult to separate from the impact of ploughing and draining. At Cray Reservoir in south Wales the cost of domestic water treatment increased markedly as a result of forest establishment with ploughing, draining and road construction all contributing to water turbity (Stretton, 1984). Similarly, sediment concentrations of up to 1200 mg l^{-1} in drainage water beside Loch Lomond in Scotland were attributed to poor drainage and road design (reviewed in Soutar, 1989).

The cost to the water industry of forest ground treatment can be very great. (Bailey-Watts, Kirika and Howell, 1988) quote costs at two reservoirs. In 1981 at Holmestyes in Yorkshire a treatment plant costing £143,035 was required, after ploughing of the catchment, to achieve a water quality in line with pre-ploughing quality. Other costs to the water industry associated with ploughing of this catchment amounted to £34,605. At Cray it was concluded that water contamination cost the water industry more than £2.5 million although of this £2.1 million was for a treatment plant which the industry intended to build but which had to be constructed earlier than planned, following forestry ground treatment. In both these cases the planning of the ground treatment failed to make adequate allowance for the unusually vulnerable nature of the soils in the catchments.

In the initial years that follow forest establishment, ground vegetation colonizes the soil that was exposed during site preparation making erosion less likely. When the tree canopy closes, ground vegetation dies because of lack of light although the ground vegetation is usually replaced by a blanket of needles which are firmly bound by tree roots. The reviews of (Moffat, 1988) and (Soutar, 1989) discuss two studies where suspended sediment yields were compared on adjacent catchments of established forest and open ground. At Plynlimon in mid Wales the forest catchment yielded 353 kg ha^{-1} yr^{-1} while adjacent pasture catchments yielded 121 and 61 kg $ha^{-1}yr^{-1}$. At Balquhidder in the southern Highlands of Scotland a partly forested catchment yielded 131 kg ha^{-1} yr^{-1} compared with a moorland catchment which yielded 381 kg ha^{-1} yr^{-1} although a subsequent paper (Johnson, 1988) gives means from 3 years sampling of 560 kg ha^{-1} yr^{-1} and 370 kg $ha^{-1}yr^{-1}$ respectively. These studies indicate that elevated levels of erosion, relative to unplanted land, continue throughout the forest rotation.

Research is now moving away from the quantification of the impact of afforestation on drainage water quality towards the study of how management can be further modified to reduce these impacts. A study of furrow erosion is being made at three sites in Scotland to determine the optimum spacing of collecting drains, and the impact of ploughing close to the contour is being compared with the impact of conventional ploughing at a site in Wales. Studies of the processes involved in the generation of sediment, colour and acidity will lead to an understanding of how widely relevant the various studies on the impact of ground treatment are. The variability of the results presented above demonstrates that different sites respond to ground preparation in different ways.

Fertilisers In plantations where weather or soil conditions slow the processes which make nutrients available for root uptake, it is necessary to apply artificial fertilisers to achieve adequate tree growth rates. The proportion of the forest estate which is fertilised each year is about 3%, most of which is in Scotland and Northern England.

The most widely used nutrients are phosphorus (as rock phosphate) and potassium (as potassium chloride) although nitrogen (as urea) is also used. Unlike areas of continental Europe, magnesium fertiliser is not required, probably because of heavy marine derived inputs in precipitation. The requirement for fertilisers depends on the species of tree being grown but in general upland mineral soils are in some cases fertilised with phosphorus, phosphorus is frequently used on shallow peat while on medium depth peat phosphorus and potassium are frequently used. Some forms of deep peat require little or no fertiliser while others require phosphorus, potassium and nitrogen. The average areas fertilised each year with phosphorus, phosphorus plus potassium and nitrogen are 25 000, 29 000 and 2000 ha respectively.

Use of phosphorus and potassium is almost exclusively restricted to the time of planting and the early years thereafter, at 6-10 year intervals until canopy closure. Nitrogen application is not normally required until about 6 years after planting although application is then usually required every 3 years until canopy closure. It seems that once trees have achieved full canopies continued growth can rely on the re-location of nutrients they already contain. Recommended rates at each application for nitrogen, phosphorus and potassium are 150, 60, and 100 kg ha^{-1} respectively (Taylor, 1986) .

It is interesting to compare these application rates with those commonly used in lowland agriculture. Data presented by Elsmere (1988), and based on a survey of a sample of 1165 farms, suggests application rates of 160, 12 and 47 kg ha^{-1} of nitrogen, phosphorus and potassium respectively on tillage crops in 1987, and 133, 5 and 23 kg ha^{-1} respectively to grassland. These, or similar amounts are applied annually. Application rates associated with pasture improvement in the uplands vary widely with soil type; typical rates used on peat soils are 50-80 kg nitrogen ha^{-1}, 30 kg phosphorus and 80 kg potassium (Newbould, 1985): these applications would take place at intervals of 5 to 7 years.

The most common method of fertiliser application in forestry is broadcast spreading from a helicopter. Although it is normal practice to leave a buffer strip of untreated ground adjacent to streams, fertiliser can reach streams by being washed down plough furrows into the forest drainage system. Fertilisers may also be washed through the soil to streams. Risk from both these routes is greatest when fertilisers are applied at planting because furrows and ditches are free of obstructions such as needle dams and vegetation, and because tree roots have not developed to utilise the entire ground area. Processes which will cause retention of nutrients on the site include uptake by trees, ground vegetation and the microbial population, and adsorption by soil particles. The slowly soluble nature of some fertilisers is also important in limiting the early losses by leaching.

In a lysimeter study on deep peat at Leadburn, central Scotland Malcolm and Cuttle (1983) found 16% of applied phosphorus fertiliser and 39% of applied potassium fertiliser lost by leaching through the peat over 3 years. However lower percentages were found where the two fertilisers were applied together. A catchment study in Glen Orchy in the southern Highlands of Scotland showed that 10% of applied phosphorus would be removed by the stream over the subsequent 3 years (Roberts, 1988). (Harriman, 1978) also studied catchments in the southern Highlands (Loch Ard and Braes of Angus) and concluded that, although the different fertilisers could be detected in the study streams for 2 to 3.5 years after application, the amounts of fertiliser reaching the streams were unlikely to be detrimental to fish production and in many situations could actually be beneficial. However he drew attention to potentially detrimental effects on relatively static water bodies such as lakes and reservoirs.

Phosphorus is of particular concern to the water industry because it is frequently the element which limits growth of algae in reservoirs. A number of papers report elevated levels of phosphorus in water supply reservoirs attributed to forestry activities (e.g. Gibson, 1976 in Northern Ireland and Greene, 1987 in western Scotland). (Bailey-Watts, Kirika and Howell, 1988) added various concentrations of phosphorus to laboratory containers of initially low phosphorus reservoir water to simulate inputs from forest fertilisation. Although they did find increased phytoplankton biomass associated with increased phosphorus concentrations, findings were complicated by factors such as water depth, dissolved organic matter and concentration of other nutrients.

Studies of fertiliser leaching usually concentrate on the movement of dissolved nutrients in stream water. However significant phosphorus export can take place with the phosphorus attached to the particulate matter transported in streams. If the particulate material forms sediments this phosphorus may be released back to the water over a long period of time.

The addition of fertilisers may cause pulses of ions other than those applied because of the ions in the fertilisers exchanging with ions in the soil. (Matzner *et al.*, 1983) working on mineral soils in Germany reported increased soil acidity as a result of fertilisation, leading to higher concentrations of aluminium in seepage water.

In future, quantities of fertiliser used in British forests are likely to be lower than at present. New plantations on poor soils and at high altitude will continue to require addition of phosphorus and potassium but increasingly the synergistic effect of growing spruce with a nurse species, for example slow growing provenances of lodgepole pine, is reducing requirements for nitrogen fertiliser. Research is currently under way to identify the value of sewage sludge as a forest fertiliser: it seems to have particular potential on dry heathland sites where the absence of overland flow makes the likelihood of stream contamination small (Bayes, Taylor and Moffat, in press). In the lowlands, except in the case of short rotation biomass crops and small areas of particularly nutrient deficient soils, fertiliser is unlikely to be required at all. In the second rotation, fertilisation is unlikely to be required even on sites which had to be heavily fertilised in the first rotation because of nutrient release from decomposing tree debris, although in subsequent rotations deficiencies may re-appear (see clearfelling section).

Pesticides

Because of the extreme toxicity of many pesticides, regulations and guidelines concerning their transport, storage and use are ubiquitous (e.g. Forestry Commission, 1989). A frequently used definition of pesticide includes liquids and granules that destroy pests and also those which offer protection against attack. Pests include insects, fungi and plants.

Pesticides are not used regularly throughout the forest rotation and therefore annual average quantities used per hectare are small compared with some forms of agriculture. The pesticides most commonly used by the forest industry are herbicides which are applied early in the rotation to give newly planted trees an advantage over existing vegetation and insecticides to control occasional outbreaks. Use of pesticides by the Forestry Commission in 1987-88, as supplied by manufacturers either as liquid or granules, amounted to just over 40 000 l/kg. This figure exludes fenitrothion used in northern Scotland (see below) but includes all other insecticides plus herbicides (FC Silviculture Division). Averaged over the entire estate this amounts to 45 ml/g ha^{-1}.

Cultivation before tree planting is frequently sufficient to retard the growth of ground vegetation to allow satisfactory growth of newly planted trees. However on some sites tree growth is inhibited by the ground vegetation competing for light, nutrients or water. In may be sufficient only to treat the vegetation around each tree or in narrow bands centred on each tree row. In these situations the risks to water quality are very small because of the large area of untreated soil where attenuation of the herbicides can take place. Bracken, because of its tendency to collapse and smother young trees, and heather, because its presence can induce nitrogen deficiency in some tree species most notably spruces, are particularly troublesome species to the forest manager. It is common therefore to attempt their complete killing using overall spraying with asulam or glyphosate on bracken and 2,4-D ester or glyphosate on heather (Williamson and Lane, 1989). While glyphosate does not taint water and has a very low mammalian toxicity all these herbicides are harmful to fish, making adherence to guidelines concerning contamination of water of paramount importance. It is also important to try to avoid spraying just before wet weather.

Lodgepole pine growing on deep unflushed peat is prone to defoliation and killing by larvae of the pine beauty moth. When control becomes necessary the most common treatment is by helicopter 'ultra low volume' spraying with the insecticide fenitrothion. (Morrison and Wells, 1981) investigated the impact of 300 g ha of fenitrothion on the water quality and fauna of Ballintomb Burn which drains a treated forest in north-east Scotland. The concentration of fenitrothion in the stream rose to a maximum of 18 $\mu g\ l^{-1}$ an hour after spraying but fell to 0.5 $\mu g\ l^{-1}$ after 24 hours. There was no evidence of disturbance to the resident fish population and although invertebrate drift increased markedly, caged insects in the stream were not killed. A common use for insecticides in the forest is the treatment of planting stock to prevent bark girdling by the pine weevil and black pine beetles in restocking sites which is very unlikely to have any impact on forest streams.

The application of herbicides can have an indirect effect on water quality as a result of the reduction in plant uptake. Feller (1989) has examined the ionic concentration of streams in two catchments in British Columbia where glyphosate was applied to kill vegetation competing with Douglas Fir *(Pseudotsuga menziesii):* he concluded that, in catchments similar to those studied, herbicide applications that reduce vegetation cover by < 5% are unlikely to have any significant effect on streamwater chemistry. However, when vegetation cover is reduced by 40% or more, significant changes in streamwater chemistry are likely and may last for 5 years or more; such application rates could have greater impacts on streamwater chemistry and nitrate fluxes than clearcutting or clearcutting followed by slashburning.

The need to use pesticides in the forest can be reduced by adopting particular forest management options, for instance the need for overall spraying against heather can be avoided by planting pine or larch instead of spruce. Although such decisions used to be made primarily on economic grounds, increasingly environmental implications are being considered. Research is also helping to reduce the need for pesticides, for instance, increasing understanding of the population biology of the pine beauty moth (Leather, Stoakley and Evans, 1987) is resulting in more effective targetting of insecticides and the use of biological control in the form of a virus against this moth is proving effective. However it is not possible to predict if a new disease to this country or the emergence of a virulent strain of an existing disease will necessitate wide new use of pesticides by the forest industry.

Harvesting

Harvesting modifies atmospheric inputs, as a consequence of canopy removal; interrupts nutrient cycling, with a reduction in plant uptake and a sudden input of debris to the soil surface; and modifies the microclimate at ground level. Together these changes can produce major impacts on water quality. Over the last 25 years there have been many studies in N America, Europe, Australia and New Zealand on the impacts of harvesting on water quality and a number of these have highlighted the increase in nutrient concentrations and fluxes, particularly nitrate, in drainage waters followings felling. The most dramatic effects were shown in the Hubbard Brook study in North America (Likens *et al.*, 1970) where nitrate concentrations reached 83 mg l^{-1} in the second year after felling. However, in this experiment, the felled trees were left on site and vegetation regrowth was prevented by the use of herbicides. Studies using more realistic conditions have also shown increases in nitrate concentrations, albeit much smaller and rarely exceeding EC guide values (e.g. Brown *et al.*, 1973; Wiklander, 1974; Tiedemann *et al.*, 1988; Sollins and McCorison, 1981). The magnitude of any increase in nitrate concentrations and fluxes seems to be controlled by the nitrogen status of the forest before felling, site fertility, forest age, felling practice, in particular whether slash is left on site, windrowed or removed (Vitousek, 1981, 1984; Wells and Jorgensen, 1979), and proportion of the catchment felled at any one time. Broadly, studies from north America, Scandinavia, Australia and New Zealand suggest that on nitrogen rich sites there will be large nitrate losses for a short period following felling, until regrowth of ground flora provide a sink for available nitrate, and small, delayed increases which then persist for several years, on nitrogen poor sites. However, the effects of felling debris varies between sites.

Although many studies had been carried out in other countries, there were no UK data until the early 1980s. Results are now available from Kershope and Beddgelert forests, and preliminary information from studies at Plynlimon, Balquihidder and Loch Ard. Following felling, drainage waters at these sites have shown increases in concentrations and fluxes of potassium, and nitrate but reductions in the mainly atmospherically derived ions, sodium, magnesium, chloride and sulphate. Ammonium and phosphate concentrations and fluxes also increased at Kershope but not at the other sites.

At Kershope where the whole of 2 ha experimental plots on a relatively nitrogen poor site were felled, weighted mean annual concentrations reached 4 mgl^{-1} NO_3-N in the second year after felling with a maximum value of 11 (Adamson *et al.*, 1987); 4 years after felling concentrations had declined again to pre-felling levels (Adamson and Hornung, in press).

Concentrations had only exceeded EC guide values on one sampling occasion during the 5-year study period and even on that occasion would be rapidly reduced by downstream dilution with water from unfelled areas. Peak fluxes of NO_3 -N at Kershope were 39 kg ha^{-1} yr^{-1} , compared with 8 kg ha^{-1} yr^{-1} in the unfelled control, but fluxes were still above pre-felling levels four years after felling due to the increased water yield. At Beddgelert, > 11 kg ha^{-1} yr^{-1} of nitrate were being leached in soil drainage waters at l m depth before harvesting and losses after felling were greater -than at Kershope with approximately 80 kg ha^{-1} yr^{-1} leaving the rooting zone in the second year following felling (Stevens and Hornung, 1988).

The greater nitrate output at Beddgelert than Kershope both before and after felling probably reflects greater site fertility (in terms of N availability) and the older crop, 55 years compared with 40 years. Concentrations in soil waters at 1 m at Beddgelert were slightly higher than the drainage waters at Kershope, with a weighted annual mean of *c.* 6 mg l^{-1} NO_3 -N in the second year after felling. However, NO -N concentrations in streams at Beddgelert only reached 1.6 mgl^{-1} , well within EC guide values. Sixty-two per cent of one catchment was felled at Beddgelert and 28% of a second; the additional nitrate leached from soils following felling was clearly diluted by mixing with low nitrate waters from the rest of the catchment. Even felling of 28% of a catchment in one operation would be unusual. At the Beddgelert experiment removal of slash resulted in lower fluxes of nitrate from soils and a rapid decline in concentrations to pre-felling levels, after 18 months. Nitrate production is apparently enhanced under brash; probably as a result of changed micro-climate conditions, while the more rapid recolonisation of the bare ground when brash is removed, reduces nitrate leaching as a result of plant uptake.

In higher N status sites than Beddgelert nitrate fluxes following felling would be even higher but for a shorter period as recolonisation would be rapid. For example, data from an experiment which involved ploughing upland grassland on brown podzolic soils showed a large flux of nitrate, with maximum concentrations of 10 mgl^{-1} NO_3 -N but the enhanced outputs only lasted one year as the site revegetated rapidly.

The increases in nitrate concentrations and fluxes are unlikely to necessitate increased water treatment but they inevitably raise background nitrate levels in the short term and reduce the 'value' of these normally very low nitrate waters in diluting more polluted waters downstream. Studies in the USA have shown that maintenance of buffer strips of woodland in the riparian zone could significantly reduce the nitrate levels in streams following felling. In Britain, where stream-side planting has taken place and wind exposure allows, it may be helpful to delay felling the immediate riparian zone until a few years after felling adjacent areas.

As noted above, increased phosphate fluxes and concentrations were measured at the Kershope site following felling of experimental plots. Outputs were, however, still low at *c.* 0.5 kg ha^{-1} yr^{-1} and were linked to periods of high flow when most water reached the drainage ditches as lateral flow through the peaty surface horizon. A large flux of phosphate was measured through the near surface soil horizons at Beddgelert following felling with brash left on site but this phosphate was retained in the deeper horizons and did not reach the streams. Soil type, water pathways and the proportion of the catchment felled are the major factors controlling phosphate output. The measured increases in output at Kershope do not represent a major deterioration in water quality but could have an impact on algal growth in small standing water bodies close to the felling. Any increase in algal growth in a supply reservoir could result in increased water treatment costs.

The mechanical disturbance of soils during felling and the greater traffic on forest roads can lead to increased suspended sediment loads. Few quantitative data are available but results from the Plynlimon catchments showed an increase in annual yields of suspended sediment from 24.4 tonnes km^{-2} yr^{-1} to 57.1 tonnes km^{-2} yr^{-2} . Bed-load also increased dramatically during felling with a five-fold increase at the catchment outlet and an increase in one small subcatcatchment from 1.2 m^3 km^{-2} yr^{-1} 1 to 23.4 m^3 km^{-3} yr^{-1} (Leeks and Roberts, 1987). Part of the increase in sediment output measured at Plynlimon will be due to erosion in a badly designed drainage system. Ferguson (1989) reported suspended sediment concentrations, in the 3 months following felling, of almost ten times those before felling; however, concentrations then declined to only 50% above pre-felling levels. It is suggested that the initial large increase could have been due to disturbance during riparian zone operations coindent with an extremely wet period of weather.

More data are needed on this aspect of the consequence of felling on water quality but if the above results are typical they would suggest that impacts on sediment yield are likely to be a greater problem than impacts on nutrient outputs, although the latter have received more attention and comment. The adoption of the practices recommended in *Forests and Water Guidelines* (Forestry Commission, 1988) should do much to minimise sediment outputs.

THE CONSEQUENCES OF AN EXPANSION IN FORESTRY

The impact of any expansion of forestry on water yield and quality will depend on the site characteristics of the planted area, the species planted and forest management. In this context, site characteristics will include climate, pollution climate, soils, geology and hydrology. The impact on any single water resource (stream, reservoir, lake, aquifer) will also depend on the proportion of the catchment or recharge area planted. To simplify discussion, possible expansion will be considered in terms of a simple physiographic division of the UK into the uplands, the upland margins and the lowlands (Figure 5). These divisions will be defined in broad soil terms; the uplands being equated with those areas dominated by hill peats, podzols, stagnopodzols, stagnohumic gleys and rankers; the hill margins as the land dominated by acid brown soils, brown podzolic soils and the associated stagnogleys; the lowlands with brown soils, rendzinas and associated stagnogleys. The results outlined in chapter 2 above will be applied in the context of these three zones. Scientific research including that described in Chapter 2 has led to changes in forest practices which were formalised with the publication *Forests and Water Guidelines* (Forestry Commission, 1988). These guidelines describe, in general terms, management actions and precautions which should be taken to maintain water quality in streams and rivers flowing through forested areas. Any future expansion of forestry in the three zones will take these guidelines into account.

Figure 5 Distribution of upland (heavy stipple) marginal land (medium stipple) and lowland (light stipple). The divisions are based on soil type and are described in section 3. Around most areas of upland there is a band of marginal land, however it is commonly too narrow to indicate on a map of this scale. The scale also prevents inclusion of small areas of marginal land and lowland within areas of upland.

Expansion in the uplands

Forestry is the most intensive form of land use, involving the greatest disturbance, in many parts of the uplands. Agriculture in this zone is generally extensive, largely involving exploitation of the existing vegetation resource with little use of fertilisers or cultivation. However, where large scale pasture improvement has taken place, the use of fertilisers will be similar to, or greater than that in a forestry rotation and some cultivation will have taken place. Management of sporting estates may also involve some drainage of peats and periodic burning of heather; the peat drainage could have a major impact on water quality.

Further forest planting in the uplands will lead to a significant reduction in water yield, largest in the wetter west. However, a review of water supplies in Scotland (Scottish Development Department, 1985) for the period 1970-2010 has included an allowance in the calculations for the reduction in yield due to additional planting and still concludes that 'developed resources are adequate to meet demands at both national and regional levels'. The review has, however, noted that a number of local supply problems remain.

(Barrow *et al.*, 1986) have examined the effect of afforestation and the consequent loss in water yield on the generation of hydroelectric power (HEP), most of which takes place in the upland zone. The authors suggest that it is important to consider each case on its individual characteristics as the 'competition' is delicately balanced but that 'where HEP capacity is already installed, forestry would ideally be diverted from the catchment'. They concluded that in discounted cash flow terms, most potential afforestation sites were unprofitable when electricity losses (as a result of a loss of water) were included and that 'Afforestation at high elevations on power generation catchments is unlikely to be justified'.

Planting on the predominantly wet, acid soils will continue to require some ground preparation and drainage with an inevitable increase in erosion, suspended sediment and colour; ground preparation of peats and peaty soils can also have impacts on water chemistry. The magnitude of these various impacts can be minimised by careful design of ground preparation schemes. Road building carries risks of erosion, especially on steep slopes but adherence to existing guidelines should reduce this to acceptable levels. Areas with deeper, unconsolidated, erosion-prone materials can be identified by good soil and site survey – the risks of erosion from even carefully planned ground preparation and road building may be so high on a small proportion of sites that planting is inadvisable.

Planting on some acid upland soils will require fertilisers. Some of the added fertiliser will inevitably be lost in drainage waters; however, the loss and consequent contamination of drainage can be minimised – the risk of loss is greater from aerial application than hand spreading, and greater on peats and peaty soils than freely drained, acidic mineral soils. Sites could be ranked in terms of 'retention capacity' for added fertilisers. In catchments of supply reservoirs with soils having a low phosphate retention capacity, the proportion of the catchment treated in any one year may have to be limited; in practice, this already happens because of the age structure of the forest estate. Weather conditions will continue to be the biggest unknown; heavy rainfall shortly after fertiliser application will increase losses.

The predicted changes in climate over the next 50 to 100 years could have a significant impact on nutrient availability in organic rich upland soils. Increased temperatures could lead to increased rates of organic matter breakdown and hence nutrient release, provided moisture does not become limiting. Such changes could reduce the requirement for fertilisers.

Applications of pesticides are likely to be very limited in the uplands. Hand spraying around individual trees has little risk of contamination of drainage waters. Greater risks attend aerial spraying in connection with pest outbreaks, but these are rare. The greatest risk is probably human error in disposal, or other contravention of approved practice.

Afforestation will increase atmospheric deposition of mobile anions and on sensitive sites this may lead to mobilisation of acidity and/or aluminium to drainage waters with adverse impacts on aquatic biota. On the most sensitive sites ameliorative liming may need to be incorporated into the forest management plan. However, atmospheric concentrations of SO_2 are declining as a result of emission control policies and this should be taken into account when assessing the impact of afforestation on atmospheric inputs and stream acidity.

The impacts of harvesting will, as in all zones, vary with site fertility, felling practice and proportion of the catchment felled. On the most acid, least fertile sites, the release of nitrate following felling will be small but may last several years. The nitrate output will be greater on more fertile sites but will only last for 3 or 4 years. The nitrate concentrations in drainage waters are unlikely to exceed EC drinking water standards even on fertile sites and the impact on lakes and impoundments downstream will be small. Phosphate outputs to streams following felling will be strongly influenced by soil type: significant quantities may be lost from peats and peat surfaced soils and could cause an increase in biological activity in downstream impoundments with risks of increased treatment costs.

(Kay and Stoner, 1988) have presented an interesting comparison of the influence of forestry and pasture improvement on compliance with EC drinking water and salmonid fisheries directives in an area of upland west Wales. Forested catchments within acid sensitive areas were associated with low compliance with the salmonid fisheries directive on dissolved aluminium and pH and pasture improvement was associated with low compliance with the drinking water directive on enteric bacteria concentrations. Afforestation of the catchments of supply reservoirs would lead to a reduction in enteric bacteria levels in the waters.

Expansion in the upland margins

In the upland margins, forestry can be viewed as a similar intensity of land use to the agriculture of this zone. The agriculture is mainly livestock oriented, but with relatively high stock densities, areas of pasture improvement and limited growth of arable crops.

Planting in these areas, dominated by brown podzolic soils, acidic brown soils and stagnogleys does not generally require extensive cultivation or drainage networks. Screef planting should be adequate on the freely drained soils to suppress weed competition. The risks of increased erosion as a result of planting are therefore small. Indeed, on some steeply sloping sites, afforestation may provide protection and a reduction in erosion. However, erosion from roads cut on steep slopes can result in considerable erosion; adherence to guidelines will minimise this risk. The absence of continuous plough furrows and drainage networks also means there is relatively little impact on site hydrology and residence times, at least until canopy closure. The soils generally have a higher base saturation than the podzols and stagnohumic gleys of the uplands; the risk of mobilisation of acidity and aluminium into surface waters as a result of any increased inputs of mobile anions is less than on the more acid upland soils but still significant and will vary with pollution climate. Choice of species can influence the magnitude of the increase in mobile anion inputs and the slightly more base rich soils and less severe climatic conditions in this zone allow more flexibility in choice of species. Hardwoods would minimise the increase while pines and spruces would produce the largest increases. The range of species that could be grown in this zone may be increased further as a result of the predicted changes in climate over the next 100 years. The nature of the bedrock and any drift deposits are important as vertical water movement dominates in the freely drained soils, and acidity and aluminium mobilised in the soils can be buffered by reactions at depth.

Planting will inevitably produce an increase in interception losses, and hence reduced water yield although the magnitude of the effect will be less than in the uplands. The increase in interception will also vary with species being higher for conifers than hardwood species.

The higher nutrient status of soils in the upland margins, compared with the uplands, also results in a lower requirement for fertilisers. If P fertilisers are used, the risk of loss to drainage waters is further reduced as the freely drained soils generally have a high phosphate sorption capacity: the poorly drained soils have lower adsorption capacities but higher than the peats and peaty soils of the uplands. The total amounts of P applied to forests are lower than those used in pasture improvement in this zone because they are applied less frequently. Nitrogen will be applied annually to any intensively used pastures or arable land in this zone.

The more vigorous growth of ground flora in the better climatic conditions, and on more fertile soils could result in a greater use of herbicides, to reduce competition, than in the uplands. The risks of contamination of surface and groundwaters is small, however, providing established guidelines are followed.

There will probably be a large release of nitrate to drainage waters at clearcutting of coniferous stands with minimal groundflora. However, the many stands in this zone would have a significant groundflora at the time of felling and, even if the groundflora were sparse, the increased output of nitrate would be shortlived as rapid vegetation spread would take place after felling. The impact of the increased nitrate release on concentrations in surface waters will depend on the proportion of the catchment felled. The contamination of groundwater is not a significant risk in this zone; the zone is not underlain by major aquifers as the majority of the rocks are impermeable. Background levels in surface waters in this zone will be higher than in the uplands and releases due to forestry activities will be similar to those due to agricultural activities such as pasture improvement, although less frequent, but less than those from more intensively grazed land or arable.

Phosphate released from felling debris will be retained in the freely drained soils and the risk of significant phosphate output to drainage waters is slight in this zone.

Expansion in the lowlands

In the lowlands, forestry as currently practised is a much less intense form of land management than the bulk of lowland agriculture. Lowland streams, rivers and lakes also have high background concentrations of solutes. Surface and near-surface run-off is also limited in many lowland areas, thus most drainage waters from the root zone are further modified by reaction with drift or bedrock before entering streams or aquifers. Water yield will be reduced following planting due to the increased evapotranspiration and the impact on streamflow and aquifer recharge needs further assessment. The reduction in water yield will be less than in the wetter uplands but may still have significant impacts on available water resource. The impact on water yield may be more significant given the reductions in annual rainfall as a result of climatic change, which have been predicted by some climatologists for southern Britain.

Ground preparation will generally be less intensive than in the uplands with, therefore, little effect on erosion and suspended sediment loads except on particularly sensitive materials. These sensitive soils are known; most are freely drained, sandy soils and the need for ground preparation would be minimal. On such soils, planting could be seen as a benefit as erosion will be much less than under some of the current agricultural systems. However, any ploughing of the nutrient rich lowland soils will lead to significant release of nitrate to surface or groundwaters until groundflora and the forest crop are established; although ploughing would, at most, only take place once per rotation of 50-60 years.

Most lowland soils in the UK have a relatively high base status and are well buffered against acidification. Thus, any increased inputs of mobile anions due to the greater deposition on to the forest canopy are unlikely to lead to mobilisation of protons and or aluminium. The areas of poorly buffered soils can be identified; the risk of acidification should be considered at these sites but there are few areas where acidic waters from the rooting zone enter streams or aquifers directly.

Fertilisers are unlikely to be required on most lowland sites and, where required the rates used will be lower than those currently used in intensive agriculture and applied much less frequently. Forestry in areas currently under intensive agriculture will reduce the output of fertiliser derived nutrients to surface or groundwaters. There are few data comparing nutrient outputs from woodlands and from intensive agriculture; however, data from Slapton, Devon (Burt and Arkell, 1987) shows annual nitrate outputs in streams from a woodland to be half those in outputs from arable and intensive grassland. Oakes *et al.*, (1981) have examined the effects of land use and management on groundwater quality at a number of sites and found concentrations of 15-50 mg l^{-1} nitrate-nitrogen below areas of arable farming but < 5mg l^{-1} nitrate-nitrogen below unfertilised grassland or woodland, and often below 1 mg l^{-1}.

Herbicide requirements may be greater than in the uplands to supress vigorously growing groundflora. Hand application will reduce risks of pollution of groundwaters. Application rates and frequencies will be much less than currently used in intensive agriculture and again forestry would be seen as less likely to cause pollution than agriculture.

The impact of harvesting will depend on site properties and method of harvest. Clear felling will have the greatest effect but the magnitude of this effect will be influenced by the amount of groundflora and the level of disturbance to the groundflora. In the absence of groundflora, clear felling will almost certainly result in large, but short-lived fluxes of nitrate, and possibly phosphate, to surface or groundwaters until a groundflora is established. Losses will be much less given a vigorous groundflora or selective felling. The increased nitrate concentrations following felling are likely to be similar to those in drainage from intensive arable and grazed grassland systems. However, clear felling would only take place once every 40-70 years and the area felled in any one year will be relatively small.

Rotation length will be another important control on the release of nutrients from forests. Release will always be less than from agriculture, but the difference will be most marked with long rotation crops. A pulse of nutrients will be released every 70-100+ years compared with annually with arable crops. Short rotation forests with clearcutting will release pulses at shorter intervals. This type of forest may also require significant fertiliser inputs with increased risks of pollution – although still low compared with intensive arable agriculture.

Atmospheric deposition forms an additional source of nitrogen which could lead to water pollution and, in some cases acidification. Planting in areas with intensive livestock enterprises or high pollutant levels from industrial/urban sources will result in large increases in deposition of nitrogen compounds because of trapping by tree canopies. On the soils with limited buffering capacity, this could lead to pronounced soil acidification and mobilization of aluminium. Nitrate pollution of waters could also result. In the Netherlands the main effects are found downwind of large-scale pig units. There are relatively few such units in the UK but planting in the intensive livestock areas of the west Midlands and south-west England will lead to sharp increases in ammonia deposition: there are insufficient data to quantify the possible increase or its impacts. The magnitude of an increase in deposition will be influenced by the species planted, being greater for conifers than for hardwoods. Croll and Hayes (1988) have calculated that, in the drier areas of lowland Britain, leaching rates of nitrate-nitrogen must be less than 20 kg N ha^{-1} yr^{-1} to ensure nitrate concentrations in the groundwater of below the EEC limits of 50 mg l^{-1}. Planting in parts of the lowlands could result in N inputs of *c.* 100 kg N ha^{-1} yr^{-1}. Present evidence suggests that leaching from the woodlands should remain below 20 kg N ha^{-1} yr^{-1} but further data are required; current work by the BGS Hydrogeology group in lowland woodlands will provide valuable additional data. This would have an important influence on decisions about further planting in 'nitrogen sensitive areas' where atmospheric levels of N gases are high. It is worth noting that riparian planting in the lowlands has been shown in North America to protect streams and rivers from runoff from agriculture (Lowrance *et al.*, 1984).

SUMMARY

This broad review of the impacts of further planting suggests that the main adverse impacts on water quality and yield will be in the uplands where forestry is a relatively intensive form of land use. The impacts can be minimised but will result in some reductions in water yield and quality. Perhaps the most likely quality problems are sediment yield and acidity/aluminium in sensitive areas.

In the lowlands, forestry as currently practised can be viewed as a more benign form of land use than intensive agriculture, especially arable. A switch from intensive agriculture to forestry would have a beneficial impact on water quality. However, the impact on water yield could be important.

If forestry is to expand in the uplands, attention should be paid to the implementation and further development of ground preparation and drainage techniques which reduce surface disturbance, soil exposure and do not produce continuous water pathways to streams. This will reduce erosion and suspended sediment load; it may also limit the production of 'colour'. Further research may also be needed on cost effective liming strategies to ameliorate any increase in acidity/aluminium consequent upon plantings: in the medium and long term however, emission abatement should reduce the need for amelioration measures.

Methods for the identification of sensitive sites should also be refined, eg materials sensitive to erosion, acidic sensitive areas, sites with low phosphate sorption capacity. The impacts of fertilisation and harvesting are controllable given careful planning; however, models are needed which will predict nitrate and phosphate release at felling and potential phosphate loss from fertilisers, for specific sites.

The broad overview can mask, however, important local impacts. The assessment of the impact of further planting should be site specific and related to the catchment or aquifer recharge area affected. The 'sensitivity' of the area to specific impacts can be ranked and could form part of the decision-making process (McDonald and Kay, 1987). Thus, small local direct supply sources (lakes or reservoirs) for remote communities are highly sensitive being influenced by reductions in water yield and quality: alternative supplies cannot readily be found and any 'pollution' will lead to major increases in treatment costs. Hydroelectric power catchments would be sensitive to reductions in water yield but less so to reduction in quality except for sediment load. Impact rating should consider both site properties, the proximity to a water source and the proportion of the catchment/recharge area to be planted.

The land use and management of the unplanted portion of the catchment or aquifer recharge zone will also influence the magnitude of the impact of forestry on water quality. Thus, any increase in aluminium or acidity resulting from planting in a sensitive upland catchment may be buffered if land elsewhere in the catchment is limed as part of a programme of pasture improvement. The impact of this agricultural liming will, however be less than a programme of liming designed specifically to buffer any increased acidity/aluminium.

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'FORESTRY EXPANSION: A STUDY OF TECHNICAL, ECONOMIC AND ECOLOGICAL FACTORS'

This is one of a series of papers which form part of a study to consider the scale, location and nature of forestry expansion in Britain.

The Forestry Commission invited fourteen specialist authors, including economists, foresters, ecologists and biological scientists to write about current knowledge and to assess the main factors bearing on decisions about the future direction of forestry expansion. It is intended that the papers will form the basis for future discussions of the location and type of forestry that will best meet the demands of society for wood products, jobs, recreation, amenity, wildlife conservation, carbon storage and the other uses and public benefits supplied by the country's forests.

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The full list of papers is as follows:

Occasional Paper No	Title	Author
33	Introduction	Professor Ian Cunningham, Macaulay Land Use Research Institute
34	British Forestry in 1990	Hugh Miller, University of Aberdeen
35	International Environmental Impacts: Acid Rain and the Greenhouse Effect	Melvyn Cannell and John Cape, Institute of Terrestrial Ecology
36	The Long Term Global Demand for and Supply of Wood	Mike Arnold, Oxford Forestry Institute
37	UK Demand for and Supply of Wood and Wood Products	Adrian Whiteman, Forestry Commission
38	Development of the British Wood Processing Industries	Iain McNicoll and Peter McGregor, University of Strathclyde and Bill Mutch, Consultant
39	The Demand for Forests for Recreation	John Benson and Ken Willis, University of Newcastle
40	Forests as Wildlife Habitat	John Good, Ian Newton, John Miles, Rob Marrs and John Nicholas Greatorex-Davies, Institute of Terrestrial Ecology
41	Forestry and the Conservation and Enhancement of Landscape	Duncan Campbell and Roddie Fairley, Countryside Commission for Scotland
42	The Impacts on Water Quality and Quantity	Mike Hornung and John Adamson, Institute of Terrestrial Ecology
43	Sporting Recreational Use of Land	James McGilvray and Roger Perman, University of Strathclyde
44	The Agricultural Demand for Land: Its Availability and Cost for Forestry	David Harvey, University of Newcastle
45	Forestry in the Rural Economy	John Strak and Chris Mackel, Consultants
46	New Planting Methods, Costs and Returns	Jim Dewar, Forestry Commission
47	Assessing the Returns to the Economy and to Society from Investments in Forestry	David Pearce, University College London

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