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The effects of forestry on soils, soil water and surface water chemistry

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Summary
The changes in soils, soil waters and drainage waters, resulting from afforestation with exotic conifers in the uplands, are discussed. The changes result partly from the replacement of the grassland, moorland or bog vegetation with forest vegetation, and partly from forest management practices. The impacts on soils are discussed for 2 groups: (i) brown podzolic soils and brown soils, and (ii) stagnopodzols, stagnohumic gleys and peats. The results of recent work on soil solution chemistry of brown podzolic soils and stagnopodzols under moorland and conifer plantations are outlined and linked to differences in chemistry of streams draining from moorland and forests. The impact of clearfelling on soil water and drainage water chemistry is reported; the impact of losses at felling on site fertility is considered. The long-term consequences of the changes in soils and soil waters, resulting from afforestation, are discussed.

1 Introduction
Vegetation is one of the major factors influencing soil properties and processes (Jenny 1941, 1980); studies in natural communities have clearly demonstrated changes in soil chemistry, biology and processes following invasion of heath or grassland by forests, or even by single trees (eg see review by Hornung 1985). The establishment of large conifer forests in upland Wales represents a drastic change in the plant cover of the planted areas. In most cases, the forests have replaced low-growing grassland, moorland or bog vegetation, but in some areas they have replaced hardwood stands. It would not be surprising if such a change in the vegetation, especially the conversion from non-tree vegetation to forest, resulted in changes to the underlying soils.

The establishment of forests in upland areas of Britain also involves site management which will influence the physical, biological and chemical properties of soils, eg ploughing, drainage, fertilization. The cycle of forest management, site preparation—planting—thinning—clearfelling—replanting, also introduces a series of perturbations into the ecosystem. These perturbations are likely to produce a cyclical pattern of variation in soil conditions which themselves may be reflected in changes in direction, or rate, of soil processes.

The physical, biological and chemical properties, and processes, of soils are also major factors influencing the chemistry of surface drainage waters. Physical properties influence water pathways, from surface to stream, and the residence time of water within the soil system. The chemical properties influence soil/water reactions, and hence the chemistry of drainage waters. Biological processes in soils often control the uptake and release of elements in soils, and to and from soil waters, and have a major impact on drainage water quality. If the creation and management of coniferous plantations cause modifications in soil properties and processes, there are likely to be consequent changes in water quality.

This paper identifies the changes in the site and soil environment consequent upon afforestation, reviews the results of relevant research, and assesses the long-term consequences of any changes. It draws upon data from recent and current research in Wales but, where relevant, also uses information from studies on similar soil types in the rest of Britain and in western Europe.

2 Changes in the site and soil environment following afforestation
Before reviewing the results of specific studies on the impact of conifers on soils, it is useful to identify those changes in the site and in vegetation/atmosphere, and vegetation/soil interactions consequent upon afforestation. We will consider 2 groups of interacting changes:
— those resulting from growth of the coniferous trees
— those resulting from forest management practices.

2.1 Changes resulting from tree growth
For the purpose of this discussion, we will concentrate on those situations where plantations have replaced grassland, moorland or bog vegetation. During the early years of a plantation, the trees probably exert little influence on the soils, and site management impacts will dominate at this stage. The main impacts of the trees probably become evident following canopy closure. In western Britain, interception losses of precipitation from the tree canopy are some 10–20% greater than those from grass or moorland vegetation (Calder & Newson 1979). Thus, 10–20% less moisture reaches the ground surface, and solute concentrations in precipitation will be increased in proportion. The tree canopy is also a much more efficient collector of particulate and aerosol material from the atmosphere than the low-growing grass, heath and bog species (Miller & Miller 1980). Conifers are more efficient collectors than broadleaves, such as oak (*Quercus* spp.) or beech (*Fagus* spp.) (Ulrich 1983). A particularly important effect in the uplands of western Britain seems to be the very efficient capture of aerosols in occult precipitation (mist, fog and cloud)

by conifer canopies (Unsworth 1984): this occult deposition has much higher concentrations of solutes, pollutant and non-pollutant, than rain or snow (Dollard *et al.* 1983; Lovett *et al.* 1982). As a result of this increased load of solutes 'captured' from the atmosphere, plus the concentration effect due to the enhanced transpiration and solutes washed from the tree canopy by canopy leaching, the waters reaching the ground below the forest canopies differ markedly in chemistry from those below grass or heath vegetation (eg Table 1). The higher concentrations of solutes in the throughfall under conifers than under grass or heath vegetation may be a more important factor influencing soil/solute interactions than the slightly increased acidity.

Table 1. Mean solute concentrations for one year in throughfall beneath Sitka spruce and mat-grass grassland at Plynlimon (concentrations in mg l^{-1}, except pH)

	Sitka spruce	Mat-grass
pH	4.4	4.6
Na	3.5	2.4
K	2.2	2.0
Ca	1.0	0.7
Mg	0.7	0.5
NH_4-N	0.28	0.15
NO_3-N	0.22	0.07
SO_4-S	2.31	0.96
Cl	5.9	5.0
DTOC*	5.3	11.8

* Dissolved total organic carbon

The amount, and chemistry, of the water reaching the ground also varies spatially. Stemflow is concentrated around the bases of the stems and can produce a large input in a small area. Stemflow also contains higher concentrations of solutes, and is usually more acid than the throughfall, or canopy drip (Table 2). In addition, the chemistry of throughfall varies markedly with tree species and age (Miller 1984a). Of the more commonly grown species, larch (*Larix* spp.) produces a more acid throughfall than spruce (*Picea* spp.) or pine (*Pinus* spp.) (eg Table 3). A study of an age sequence of Sitka spruce (*Picea sitchensis*) in Beddgelert Forest showed that up to 35 years of age throughfall was less acid than the incoming precipitation, but in older crops it was consistently more acid (Stevens 1987).

The rooting pattern of trees is very different from that of the grass, heath or moorland vegetation they replace; rooting patterns will vary considerably with species and soils. Tree roots create many large pores which, in drier soils, provide pathways for water movement. The increased transpiration of trees, added to the reduced water input to the soil, can result in a significant drying and cracking of soils. The rooting pores and drying cracks facilitate rapid water movement and can significantly alter the hydrological properties of the soils; they also provide avenues for air entry into the soil. The drying of the soil will result in

Table 2. Volume-weighted mean solute concentrations for one year in throughfall and stemflow beneath Sitka spruce (P.1936) in Beddgelert Forest (concentrations in mg 1^{-1}, except pH)

	Throughfall	Stemflow
pH	4.07	3.62
Na	12.3	32.0
K	1.4	3.7
Ca	1.0	3.2
Mg	1.5	4.3
NH_4-N	0.65	0.62
NO_3-N	0.92	1.17
SO_4-S	3.1	7.7
H_2PO_4-P	0.027	0.036
Cl	21.3	60.5
DTOC*	2.0	7.0

* Dissolved total organic carbon

changes in biological and chemical processes; the presence of roots in deeper horizons may also lead to changes in weathering and element release.

The organic matter on the forest floor may be very different in type, biology and chemistry to that from grass or heath vegetation. Most coniferous species will produce a moder, or mor humus, which may be significantly more acid than the organic horizons, if any, produced by the preceding vegetation. The breakdown products of the humus will also differ from those of other humus types. Nutrient cycling in the forest is markedly different to that in the preceding non-tree vegetation. In the early years of a plantation, there is a net accumulation of elements in the tree and the forest floor (the organic horizons which accumulate below the forest). At later stages, the release of nutrients from the forest floor, by decomposition, plus inputs from the canopy, may balance uptake (Miller 1984b). The pattern of element uptake may also differ in the forest as compared with the non-tree vegetation. While most elements will be taken up from near-surface horizons, there will be some uptake from deeper soil layers.

Table 3. Mean solute concentrations for one year in throughfall beneath Sitka spruce (P.1949) and Japanese larch (P.1949) at Plynlimon (concentrations in mg l^{-1}, except pH)

	Spruce	Larch
pH	4.37	3.9
Na	3.5	3.0
K	2.2	1.0
Ca	1.0	1.0
Mg	0.7	1.0
NH_4-N	0.28	0.15
NO_3-N	0.22	0.15
SO_4-S	2.3	3.4
Cl	5.9	6.4
DTOC*	5.3	8.2

* Dissolved total organic carbon

The dense canopy produced, particularly by spruce, will clearly result in reduced light penetration and a changed microclimate. At canopy closure, the ground flora will be shaded out, resulting in a large addition of dead plant material to the soil surface; breakdown of this material will release elements which may be taken up by the trees, held in soil or lost to drainage waters.

The temperature regime at ground level will change, with lower summer temperatures but higher winter ones than in the preceding non-forest vegetation. The changed humus and microclimate result in a modified soil fauna, with consequent influences on decomposition and mixing of surface organic materials.

2.2 Changes due to forest management practices

2.2.1 Site preparation

Most of the wetter and/or heather-dominated upland sites are ploughed prior to planting (Thompson 1984). Ploughing is designed to suppress competing vegetation, improve surface drainage and aeration, provide an improved planting position and increase rooting depth (see Plate 6). It produces inversion and mixing of surface soil. The disturbance and drying can produce an increased rate of decomposition, with an enhanced release of elements: these released elements may be taken up by the ground flora, retained within the soil or leached out into drainage waters, with a consequent impact on water quality. The ploughing also exposes subsurface soil, which may lead to increased rates of weathering and element release.

Drains are designed to remove surface and near-surface water from the site, and, therefore, to reduce waterlogging and improve rooting conditions (Thompson 1979). The drainage will cause drying of the soil and will expose subsoil horizons or drift in the ditch walls to erosion or weathering. Together, the plough furrows and drains can radically alter water pathways between soil surface and stream channel, reducing residence times of water and consequently water chemistry. A major effect may be an increase in the proportion of water reaching the streams from the surface horizons of soils.

2.2.2 Fertilization

The use of fertilizers will increase the soil store of the added elements. As the fertilizer breaks down, the released elements will be taken up by the trees or ground flora, retained in the soil or leached to drainage waters. The fertilizer may also change soil pH, leading to changes in soil processes. Phosphate, applied as rock phosphate, is the most widely used fertilizer in Welsh forests; it will add calcium to the soil, in addition to phosphorus.

2.2.3 Felling

Clearfelling, and to a lesser extent thinning, will cause a series of changes in site conditions, many of which are reversals of those associated with canopy closure (Hill *et al.* 1984). Capture of elements from the atmosphere, interception and transpiration loss will be reduced. The input of elements to the soil surface will

Plate 6. Land ploughed prior to planting—Dyfi forest (Photograph J H Williams)

Plate 7. Brash produced when felling the first rotation—Helmsley Forest, Yorkshire (Photograph J H Williams)

decline, but water inputs will increase. There is a sudden large addition of plant material to the soil surface as slash (see Plate 7). Uptake by plants of nutrients released by decomposition, and from mineral sources, will effectively cease. Light penetration and the microclimate at ground level will change. A significant quantity of nutrients will be removed from the site in the timber.

3 The impact on soils

3.1 Brown earths and brown podzolic soils

Most of the early, extensive, planting of exotic conifers in Wales was on brown earths and brown podzolic soils of the lower valley-sides. These soils would have carried mixed oak/birch (*Betula* spp.) woodland prior to forest clearance and would have largely evolved as forest soils. There has been little work in Wales on the impact of coniferous plantations on these soils, but a great deal of research on broadly similar soils (sol brun acide, sol brun ochreux, sol brun podzolique) has been carried out in western Europe.

Most of these studies compare the soils beneath adjacent stands of hardwoods and conifers, or below hardwood and conifer stands on soils which are assumed to have been similar prior to planting the conifers. Clearly, great care must be taken in this type of study to ensure that detected soil differences are a result of the differences in vegetation and not due to inherent variations in the soils. Because of this kind of consideration, many early studies which report acidification and podzolization of soils by conifers are now questioned (Stone 1975). There are, however, a number of reliable recent comparative studies, based on carefully evaluated sites, plus process studies comparing hardwood and conifer systems. Taken together, these studies identify a variety of soil changes resulting from growth of monocultures of exotic conifers, especially spruce. The most consistently reported change is the development of an acid mor humus, replacing mull or moder, with a lower nitrogen content and higher carbon/nitrogen ratio (eg Nihlgard 1971; Herbauts & de Buyl 1981; Nys & Ranger 1985). One result of the change in organic matter is an increased proportion of fulvic to humic acids in the breakdown products. The surface and near surface horizons are also generally acidified and the content of both total and exchangeable base cations reduced. The loss of base cations is generally assumed to indicate increased mineral breakdown and leaching, but Nys and Ranger (1985) concluded that the potassium and magnesium losses from the surface horizon which they measured under spruce were a result of increased clay illuviation; this clay was retained in the B horizon and did not represent a loss from the site. An increased translocation of iron and aluminium from near-surface horizons has been reported from a number of studies (eg Nys & Ranger 1985; Herbauts & de Buyl 1981); in a few instances, the development of a clear eluvial horizon has been reported (Nihlgard 1971; Bonneau 1973). Herbauts and de Buyl (1981) suggest that the increased ratio of iron to aluminium in NaOH/Na-tetraborate extracts from the B horizon below spruce shows evidence of marked podzolization, compared to below beech. The same authors report an increased penetration of fulvic acids into the B horizon. Increased breakdown of clay minerals under spruce is also reported by Nys (1981) and Herbauts and de Buyl (1981); this evidence is taken as an indication of an increased trend towards podzolization. Structural changes were found by Nys and Ranger (1985), Schlenker *et al.* (1969) and Bonneau *et al.* (1977). Nys and Ranger report a 30–40% decrease in porosity beneath spruce and a reduction in structural stability. These latter authors also summarize the detected changes beneath coniferous species and assess whether or not they are reversible (Table 4).

Table 4. Summary of detected changes in soils beneath coniferous species replacing broadleaves (source: Nys & Ranger 1985)

PHYSICAL	
Bulk density change	Reversible?
Structural degradation	Reversible?
Reduction in porosity	Reversible?
Particle migration	Irreversible
CHEMICAL AND PHYSICO-CHEMICAL	
Structural degradation	Irreversible
Organic matter: loss of N	Reversible?
Reduction of mineral cation exchange capacity associated with leaching	Irreversible
Change in the exchange complex	Irreversible
Total loss of elements	Irreversible
— by drainage out of the ecosystem	Irreversible
— beyond the rooting zone	Irreversible

The impact of conifer plantations will, however, vary with soil type and with crop species. Bonneau *et al.* (1979) stress the importance of variations in soil parent material, and suggest that the impact on fine-textured parent materials, and those with high levels of exchangeable bases, is negligible. Soils with parent materials derived from crystalline rocks or acid sediments poor in iron and bases are considered the most sensitive. In this respect, it is worth noting that the development of a thick eluvial horizon under spruce, reported by Bonneau (1973), occurred on freely drained, sandy, base-poor material. Care must be taken, therefore, when extrapolating results from one site to another.

Three British studies have examined the impacts of conifers on brown earths or brown podzolic soils. Results from a study by Grieve (1978) broadly parallel those reported from France and Belgium. He compared the soils beneath stands of 50-year-old spruce with those below mixed oak/beech, last replanted in 1815, in the Forest of Dean. The soils were classified as brown earths of the Neath series. There was significant leaching of iron under the spruce, with the formation of eluvial and illuvial horizons. A discrete humus horizon had also formed under the spruce, horizon boundaries were sharper and the structure was coarser and weakened. Grieve (1978) concluded that there had been a change in the balance of the soil-forming processes with a move towards podzolization under the spruce.

Hornung and Ball (1972) compared the surface horizons in 30-year-old Sitka spruce plantations, and adjacent fescue/bent (*Festuca/Agrostis*) grasslands at 3 sites in north Wales. The soils at all 3 sites were brown podzolic of the Manod series (Rudeforth *et al.* 1984). The 0–5 and 5–10 cm depths were compared, having first removed the forest floor. Both sampled zones were significantly more acid below the spruce at all 3 sites. Two sites showed increases in loss-on-ignition and total nitrogen, whilst the third showed a reverse trend. Data on exchangeable cations showed no consistent trends. Page (1968) examined soils under size sequences of Sitka spruce, Douglas fir (*Pseudotsuga menziesii*) and Japanese larch (*Larix leptolepis*) in Gwydyr Forest, north Wales: all 3 species were growing on brown podzolic soils. Generally similar trends were found under all 3 species: litter thickness and moisture content increased up to a top height of 18 m (60′) to 27 m (80′) and then declined again to the level found in unplanted soils. Bulk density and pH decreased until a top height of 18–27 m and then increased back to the starting value below 36 m (120′) tall trees. The A_2 (Ea) horizon below Sitka spruce showed a gradual and continuous increase with top height. This study by Page (1968) highlights the difficulties in drawing conclusions from studies based on one point in the crop rotation.

Several of the changes in soils discussed above could be taken as indicating an increasing tendency towards podzolization, which may actually be an intensification of already existing processes in the soil. Thus, Herbauts and de Buyl (1981) conclude that the soil under beech shows indications of podzolization which become clearer under spruce.

3.2 Peats, stagnopodzols and stagnogleys

The majority of the extensive forest plantings in upland Wales have occurred on humic gley soils, stagnohumic gleys, stagnopodzols and peats. In some areas, particularly at the lower end of their altitudinal range, these soils have evolved from former forest soils, under the influence of heath or moorland vegetation. The replacement of the forest vegetation led eventually to the development of soils with organic, or peaty surface horizons, and zones of intensive weathering and eluviation in the near surface mineral horizons. Podzols were developed on rather better drained sites, with stagnohumic gleys or peats on benches or gentle slopes. They are all naturally very acid, base-poor soils; in the mineral horizons, the exchange complex is dominated by aluminium. There have been very few studies of the impact of afforestation on the properties of such soils. There are few areas of similar soils in mainland Europe, particularly those areas where exotic conifers have been planted. The emphasis in Britain has been on developing improved silvicultural methods for use on these rather difficult soils, or on the performance of the trees, rather than on impacts of the trees on the soils.

There have, however, been a number of studies on the drying of peat, following afforestation, and the linked changes in exchangeable cations, nitrogen availability and acidity (eg Binns 1979; Pyatt 1976; Williams *et al.* 1978; Boggie & Miller 1976). Pyatt (1976) found that the extent of drying and the development of shrinkage cracks in peat were related to the original depth of peat, degree of humification, tree species and, probably, climate. In common with other workers, he found that lodgepole pine (*Pinus contorta*) produced by far the greatest drying effect of the commonly planted species. The drying tended to be more intense and cracking more rapid in relatively shallow, and well-humified peat: at a site near Dumfries, shrinkage cracks had already developed under 9-year-old lodgepole pine. Pyatt (1976) also reports that the drying is irreversible. The increased drying below lodgepole pine, compared with the adjacent unplanted bog, increased the percentage air volume down to 40 cm, doubling it in the top 20 cm, at the site studied by Boggie and Miller (1976). Binns (1979) and Williams *et al.* (1978) report increased acidity of the dried peat, increased bulk density, increased exchangeable sodium but decreased calcium. The calcium decrease seems to be linked to uptake by the crop. The increased sodium is thought to be due to enhanced aerosol capture on the canopy. The increased acidity is said to be a result of increased exchangeable hydrogen on newly created exchange sites, formed as a consequence of greater organic matter breakdown in the more aerobic environment.

Similar changes might be expected in the surface peaty horizons of stagnohumic gley soils (peaty gleys) and stagnopodzols (peaty podzols). Pyatt (1973) has reported pronounced drying of the peaty horizon of a stagnohumic gley in Kielder Forest, Northumberland. More surprisingly, he found significant drying of the underlying mineral horizons down to 90 cm, although rooting was limited to 20–30 cm. Further studies in Kielder Forest (King *et al.* 1986) have shown considerable lowering of the water table, especially in the summer months, by Sitka spruce and lodgepole pine on stagnohumic gley soils, and a corresponding improvement in the oxygen regime. In a study of water and oxygen regimes in a range of soils in Newcastleton Forest, south Scotland, Pyatt and Smith (1983) found healthy roots present in brown earths to a depth of 85 cm and in an ironpan stagnopodzol to 75 cm. In the latter soil, the roots had penetrated the iron pan and the resultant macropores must have had an important effect on water pathways and movement in that soil. Recent work by the authors on stagnopodzols in Hafren and Towy Forests revealed an increase in cation exchange capacity and exchangeable hydrogen in the surface peaty horizon, as compared with similar soils below moorland.

Most sites with peat, stagnogley, gley or stagnopodzol soils have been ploughed and drained prior to planting. The drying effects reported from these soils will reflect the combined influence of increased interception and transpiration and of ploughing and drainage.

Ross and Malcolm (1982) examined the physical effects of ploughing on a peaty ferric stagnopodzol in south-east Scotland. The cultivated soil had lower bulk density, was better aerated, showed faster infiltration and had higher mean annual temperatures than untilled soil. The ploughing produced an intimate mixing of the organic and mineral horizons to a depth of 60 cm.

It is very surprising that there are no published data on the impacts on soil biology and chemistry of such ploughing and drainage operations.

3.3 Harvesting effects on soils

At harvesting, heavy machinery is used for felling and timber extraction. The ground traversed by this machinery becomes compacted, especially those areas used as skid trails or for repeated access by forwarders. A recent study of a site in Scotland indicated that some 10% of the ground was affected by passage of machinery (H G Miller, pers. comm.). There are, as yet, no British data available on the impact of this traffic on bulk density or structure, or on the consequential effects on infiltration and soil erosion, or on the growth of the next crop. Studies in north America and New Zealand, however, have reported significant increases in bulk density, which persisted well into the next rotation.

4 The impact on soil water and drainage water chemistry

4.1 Site preparation

As noted above, ploughing and drainage of sites involve disturbance and mixing of surface soil, and exposure of soil to the atmosphere, resulting in drying of the soil and consequent modification to soil chemical and biological processes. To date, there are no published studies on the impact of ploughing and drainage on soil water chemistry, although work is in progress by the authors as part of the Llyn Brianne study and at other sites in Wales. The soil changes produced by ploughing and drainage may be expected to influence the chemistry of surface waters draining from the site. There have, however, been few attempts at characterizing, or quantifying, any changes in water quality. Robinson (1980) monitored water quality for a short period during ploughing and drainage of a site on stagnohumic gleys and peats in Cumbria, also drained by the Coal Burn, a tributary of the Irthing. Concentrations of calcium, magnesium, nitrogen and potassium increased following ground treatment. There was also a change in the relative abundance of the 4 main cations, from Na Ca Mg K before drainage to Ca Na Mg K after drainage. The additional calcium and magnesium were probably derived from the glacial till exposed in the drains and some of the plough furrows. The increased levels of nitrogen and potassium probably reflect enhanced release from organic sources. Studies in Finland (eg Hynninen & Sepponen 1983) have found increased levels of ammonium and nitrate in drainage waters following ploughing and drainage of peats; these increases have been linked to higher rates of decomposition following drying of the peat.

Unpublished results from a study at Nant-y-Moch in west Wales show large increases in sulphate, ammonium and aluminium following ploughing and drainage (A S Gee, pers. comm.). The sulphate and ammonium were probably produced by enhanced decomposition and oxidation as a result of drying, while the aluminium was probably mobilized from the exposed mineral horizons. At Llanbrynmair in mid-Wales, ploughing close to the contour has been used in an attempt to limit the impact of site preparation on water quality. Data obtained from a study carried out at this site by the Institute of Hydrology (Leeks & Roberts, this volume) have shown no changes in drainage water chemistry following site preparation. The impacts on the soils will remain the same, but the products of the changed decomposition and the drying are being kept on-site.

4.2 Established crop

4.2.1 Soil waters

The emphasis of most studies on stagnogley soils, stagnopodzols and peats, discussed above, has been on physical changes, with only a limited amount of work on changes in peat chemistry and none on the chemistry of the deeper horizons. The authors have recently established a number of studies on soil solution chemistry at a series of sites in Wales and northern England (Hornung *et al.* 1986a, b; Reynolds *et al.* 1987). The results to date show clear differences in solution chemistry between waters extracted from initially similar soils below coniferous plantations and nearby grassland or moorland. There are also differences between the waters from different soils within the forest, and from below different crop species on the same soil.

The forest soil waters contain higher concentrations of most measured solutes than those from the moorland soils (Table 5). The largest increases are shown by sodium, chloride, sulphate and aluminium: aluminium and sulphate concentrations in the mineral horizons of the forest soil waters are almost 3 times those in the moorland soil, while sodium and chloride concentrations are some 50% higher. The aluminium appears to be mobilized from cation exchange sites, by ion exchange. Hydrogen ions, either input in throughfall or stemflow, or mobilized by ion exchange in the very acid, surface organic horizons, exchange for and displace the aluminium into solution. The process is

Table 5. Mean soil water solute concentrations for one year in the E, B and C horizons of a stagnopodzol under Sitka spruce and mat-grass/fescue grassland at Plynlimon (concentrations in mg l^{-1}, except pH)

	Forest			Grassland		
	E	B	C	E	B	C
pH	4.1	4.3	4.5	4.3	4.4	4.7
Na	4.9	6.2	5.8	4.3	4.0	4.5
K	0.1	0.2	0.3	0.2	0.2	0.3
Ca	1.1	0.9	0.7	0.4	0.3	0.3
Mg	0.7	0.7	0.6	0.5	0.5	0.6
NH_4-N	0.01	0.02	0.02	0.02	0.02	0.02
NO_3-N	0.15	0.02	0.22	0.54	0.39	0.61
SO_4-S	2.6	3.3	3.0	1.4	1.1	1.2
Cl	8.3	11.6	10.0	7.4	6.5	7.2
DTOC*	11.8	6.1	4.1	6.0	3.7	2.3
Fe	0.30	0.05	0.01	0.07	0.04	0.02
Al	1.2	1.5	1.5	0.5	0.4	0.6
Si	1.5	1.5	2.0	1.2	1.0	0.9

* Dissolved total organic carbon

driven, however, by the increased input of inorganic anions, mainly sulphate, to the forest soils (Reynolds *et al.* 1987), which are derived, ultimately, from atmospheric sources. The greater sulphate input to the forest soils reflects enhanced deposition of aerosols on to the forest canopy, with deposition of occult precipitation likely to be particularly important. There will, however, also be a contribution from the increased chloride input and from organic anions derived from the forest floor and from decomposition of the pre-existing peat horizon. The contribution from organic anions, however, seems to be small.

Aluminium concentrations in the waters from the forest soils under Sitka spruce at sites in Towy, Hafren and Beddgelert Forest range from 0.7 mg l^{-1} in brown podzolic soils to 1.0 mg l^{-1} in intergrades between brown podzolic soils and stagnopodzols, and to over 2 mg l^{-1} in an ironpan stagnopodzol at high altitude. Preliminary data also suggest a species effect, with higher aluminium concentrations in similar intergrade soils under larch (1.5 mg l^{-1}) than Sitka spruce (1.0 mg l^{-1}). This finding may reflect a greater production of nitrate in the larch forest floor (cf Carlisle & Malcolm 1986); some of the nitrate is apparently leached down the soil profile, increasing the anion load and leading to additional aluminium mobilization.

4.2.2 Surface waters

Recent studies in Wales (Stoner *et al.* 1984; Stoner & Gee 1985), north-west England (Bull & Hall 1986) and Scotland (Harriman & Morrison 1982) have reported differences in water chemistry between streams draining from established upland plantations and from adjacent unplanted moorland on otherwise similar sites. A number of solutes are present at higher concentrations in the forest streams than in those draining moorland. Concern about the differences in water chemistry has focused, however, on the increased acidity and higher aluminium concentrations in the forest streams. For example, mean annual aluminium concentrations in forest streams at our site at Beddgelert (Table 6) are greater than 0.5 mg l^{-1}, but normally less than 0.2 mg l^{-1} in a nearby moorland stream; the pH of the forest stream is 0.4 units lower than the moorland stream. These particular changes in chemistry have been linked to reductions in the diversity of invertebrate populations and reductions in fish numbers (Ormerod *et al.*, this volume).

Table 6. Discharge-weighted mean solute concentrations in streams draining Sitka spruce forest and moorland at Beddgelert (concentrations in mg l^{-1}, except pH)

	Forest	Moorland
pH	4.38	4.80
Na	6.2	4.1
K	0.25	0.12
Ca	1.5	0.9
Mg	0.9	0.6
NO_3-N	0.63	0.08
SO_4-S	2.4	1.4
Cl	11.5	7.2
Al	0.56	0.2

The link between increased aluminium concentrations and acidity, and afforestation is not found at all sites, and it seems to be restricted to those areas where acid soils overlie massive, base-poor bedrock. These conditions obtain over a large proportion of the Welsh uplands. Given these conditions, aluminium concentrations and acidity may be greater in the forest streams at all flow levels. The differences between aluminium concentrations in forest and moorland streams are, however, greatest during periods of high flow. Thus, at Beddgelert, aluminium concentrations have reached more than 2.5 mg l^{-1} during flood events, while remaining below 0.2 mg l^{-1} in adjacent moorland streams. The magnitude of the difference in solute chemistry between forest and moorland streams seems to vary somewhat with soil type, being less in catchments dominated by brown podzolic soils than in those dominated by stagnopodzols and stagnohumic gleys. The levels reached by aluminium concentrations during a specific storm vary with the intensity of the storm and with antecedent conditions. Even when streams show similar acidity, the aluminium levels may differ. Thus, Reynolds *et al.* (1986) found that forest and moorland streams on similar bedrock, in the Plynlimon area, had similar pH and calcium levels (Table 7), but that the aluminium concentrations were significantly greater in the forest stream.

The causes of the increased acidity and aluminium concentrations are still being investigated, and it seems that a number of processes may be involved (Miller 1985). The increased interception and transpiration of the forest, compared to moorland or grassland, produce a concentration effect. The deposition, or capture, of aerosols, especially those in occult deposition, is greater on to forest canopies than on to

Table 7. Discharge-weighted mean solute concentrations in streams draining Sitka spruce forest and moorland at Plynlimon (concentrations in mg l^{-1}, except pH)

	Forest	Moorland
pH	4.7	4.8
Na	4.5	3.7
K	0.16	0.14
Ca	0.9	0.8
Mg	0.8	0.7
NO_3-N	0.42	0.21
SO_4-S	1.5	1.2
Cl	7.7	6.3
Al	0.28	0.1

grassland or moorland vegetation. Together, these mechanisms produce an increased input of salts to the forest soils. The acid organic layers which develop below the forest also produce organic acids during decomposition. Uptake of nitrogen in ammonium form, by the conifers, will result in the balancing release of hydrogen ions from the roots. Accumulation of base cations in the tree may also have a long-term acidifying effect on the soils (Nilsson *et al.* 1982).

Analysis of our data on soil water chemistry suggests that ion exchange, driven mainly by the increased input of anions resulting from enhanced 'capture' on the canopy, explains the raised levels of aluminium in the forest soil waters (Reynolds *et al.* 1987). The soils provide, therefore, a source of water with high concentrations of aluminium. The ploughing and draining, and resulting drying cracks, plus the macropores created by tree roots, alter the soil and site hydrology. A major result of the changed hydrology is an increase in the proportion of water reaching streams from the acid, upper soil horizons. The water is also transferred to the stream quicker, with less time, therefore, for buffering. Whitehead *et al.* (1986) have used hydrochemical models to demonstrate the importance of changing the proportions of streamwater derived from different sources, in particular the balance between groundwater and acid surface soil water. These 2 processes of increased solute loadings in the soils, but especially of sulphate, and changed site hydrology probably interact to give the detected changes in streamwater chemistry. At our Beddgelert Forest site, however, the forest streams are more acid and contained higher concentrations of aluminium than the moorland stream (Table 5), even though the site was not ploughed or drained. At this site, the dominant influence appears to be increased mobilization of aluminium as a result of the greater anion loading.

4.3 Clearfelling

Many studies on the impact of clearfelling on water quality have been carried out in north America (eg Likens *et al.* 1970; Aubertin & Patric 1974; Cole *et al.* 1975; Vitousek & Melillo 1979), Scandinavia (eg Tamm *et al.* 1974; Haveraaen 1981) and New Zealand (eg Neary 1977; Dyck *et al.* 1981). Several of these studies have reported increased concentrations of nitrate and base cations following felling. The increased nitrate concentrations, in particular, have given rise to considerable concern about the possible effects on drinking waters. A series of studies to examine the impact of felling, in UK conditions, on soils, soil waters and water quality is now being carried out by the Institute of Terrestrial Ecology, the Institute of Hydrology, and the Forestry Commission at Beddgelert Forest, in north Wales, Kershope Forest, in Cumbria, and Hafren Forest, in mid-Wales.

Initial results from Beddgelert Forest show large increases in concentration and flux of inorganic N, potassium and phosphate-P in waters in stagnopodzol soils following felling. The nitrogen flux was greater in the second year after felling than in the first: 96 kg ha^{-1} of inorganic N was transferred below the rooting zone in the second year. This large flux of nitrate through the subsoil also resulted in an acidification of the soil waters. The major post-felling flux of potassium was in the upper horizons in the first year, and the lower horizons in the second year. In the second year, potassium concentrations in waters from the upper horizon had reverted to pre-felling levels. It appears that a front of potassium gradually moved down through the soil, taking some 2–3 years from felling to pass from the O to the C horizons. Phosphate fluxes declined in the second year after felling, but were still much greater than before felling, but little phosphate reached the subsoil, most being immobilized in the upper horizons.

Process studies at Kershope and Beddgelert Forests have confirmed that the increased K and PO_4-P fluxes after felling are derived from the felling debris. The inorganic N, however, is derived from the pre-existing forest floor and soil organic horizons. Mineralization of organic N continued after felling but, in the absence of root uptake, the nitrate produced was lost in drainage waters.

Drainage water from the Kershope site showed marked increases in concentrations of nitrate, ammonium and potassium following felling, reflecting the increased concentration in the soil solution. A parallel reduction in the levels of Na, Ca, Mg, SO_4-S and Cl in drainage water is explained by reduced 'capture' from the atmosphere and dilution, and by the increased water throughput. In the year following felling at Kershope, output of inorganic N and of K from a completed cleared plot was 5 times greater than that of an unfelled control plot. Streamwater concentrations of K and NO_3 have increased to a lesser extent at Beddgelert, where only a proportion of the catchments were felled, than at Kershope. The Beddgelert situation of partial felling of a stream catchment seems more realistic in practice than that at Kershope.

5 Long-term consequences and management options

It seems clear that monocultures of conifers, particularly spruce, cause modifications to the biological, chemical and physical properties of some brown earths, brown podzolic and similar soils. These changes are most marked on coarse-textured, base-poor parent material; Bonneau (1975) reports the development of a podzol with a distinct Ea horizon in such material. The changes include the development of a mor humus, surface acidification and leaching, an increased trend towards podzolization and structural degradation. While coming to similar conclusions, Bonneau (1978) and Manil (1966) suggest that, in most cases, the modifications to the soils are slight, most of which could be corrected by additions of fertilizer and lime, and do not represent a significant decrease in long-term site fertility. Bonneau (1978) does, however, suggest that there may be significant short-term effects on availability of some base cations, and Nys and Ranger (1985) have identified several soil changes which they regard as irreversible. The views of Manil (1966) and Bonneau (1978) are largely addressed to the continued use of the site for forestry. The mor humus, and the general surface acidification, may affect the vegetation which would develop on the site after felling; certain acid-sensitive species may not return. The magnitude of this effect will vary with the initial soil type and the tree species planted. The major control on post-felling vegetation would probably be the nature of the available soil seed bank and of nearby seed sources (cf Hill, this volume).

Structural degradation may have important consequences. At clearfelling, the exposed soil would be much more sensitive to further structural damage due to the passage of heavy equipment and raindrop impact, and, as a result, to soil erosion. On sloping sites, it may be necessary to protect the soils or to limit the size of the felling coupe. Nys and Ranger (1985) have identified most of the soil changes consequent upon planting of exotic conifers as reversible. We are unable, however, to forecast timescales for the recovery, or, more properly, the adjustment of the soils to a subsequent changed vegetation. It may take a considerable period to restore soil structure, during which time the site will be at increased risk to soil erosion.

The main effects on the peats, stagnopodzols, stagnogleys and gleys are probably physical. The combined effects of ploughing, drainage and tree growth produce a marked drying in many soils, the extent of the drying depending on initial soil conditions, ground treatment and tree species. The drying also produces oxidation of organic nitrogen and sulphur compounds and more rapid decomposition. There do not appear to be large changes in chemistry of the mineral horizons, but there are important changes in soil water chemistry with large increases in aluminium levels. Some of the drying effects, particularly in peats, are irreversible. The penetration and disruption of the ironpan in ironpan stagnopodzols and the general cracking of the E horizons in stagnopodzols may also have long-lasting effects. Some sites, especially stagnogley and gley sites, seem to 'wet-up' again remarkably quickly after felling (eg Pyatt *et al.* 1985), even though drains remain open and active. Some of the initially wetter peat sites would not, however, return quickly to the wet bog or moorland vegetation which existed before planting. The dried, humified and cracked peat may also be liable to erode following removal of the protection of the forest canopy. Base cation availability in the organic horizons has declined on some sites, but this may be a short-term effect until the forest nutrient cycle is established. It is unlikely that planting will have reduced the site fertility on these soil types, although losses of phosphorus at harvesting may be significant in the short term. There may also be problems of synchronizing nutrient availability with nutrient demand in successive rotations. The effects on any subsequent, non-forest, vegetation will most probably be linked to the drying, and physical changes, rather than to alterations in soil chemistry.

It is difficult to conceive of management options which will ameliorate the physical impacts. The establishment of a crop on the wetter soils requires some form of drainage, and the ironpan in ironpan stagnopodzols needs to be disrupted to improve rooting conditions. The drying effect due to tree growth is inevitable. The effects of ploughing and drainage on drainage water chemistry can be ameliorated by modifying the design of the ground treatment. The fact that no significant changes in solute concentrations have been detected during the second phase of ground preparation at Llanbrynmair suggests that the approaches now being used are successful. The increased ion input, especially sulphate, to forest soils, due to canopy scavenging, will not be affected by forest management. Enhanced levels of aluminium in soil waters of forests on sensitive soils, therefore, seem inevitable, without a reduction in anion inputs from the atmosphere. Even given a reduction in sulphur and nitrogen levels, the additional capture of sea-derived solutes by forest canopies, compared with moorland or grassland, will lead to higher aluminium levels in the forest soil waters, although the difference would be much less than at present. The impacts of established plantations on the acidity and aluminium concentrations in streams may be reduced by changes in design of drainage schemes. If waters can be kept on site longer, and ditches not fed directly into streams, then some additional buffering may take place. These principles are now being incorporated into new planting schemes. Liming of forest soils would also reduce acidity and aluminium levels, but it may prove to be easier to treat the problem in drainage waters than in the soils. The liming of both waters and soils is, however, now being explored. It has also been suggested that increased buffering of drainage waters would be achieved if the water moved to depth and came into contact with more base-rich subsoil, drift or rock. This approach may be feasible on some sites and could be achieved by the use of very deep drains or specially excavated sumps. The technique does not, however, seem to have much potential on the very acid slates and mudstones of central Wales.

The longer-term impacts of soil compaction produced during harvesting are as yet unknown. In the United States, reduced growth in the succeeding rotation has been reported of trees planted in the compacted soils. The affected trees tended to catch up in the later years of the rotation, and, in the relatively long rotations used in Britain, compared with the southern USA, this timescale may be acceptable. The impact should, however, be quantified. In the immediate post-felling period, the most important effects will be a reduction in infiltration, increased surface wetness and soil erosion. Some impact on water quality at clearfelling also seems inevitable. Our work at Beddgelert suggests, however, that felling of the normal-sized coupe is unlikely to produce changes in stream chemistry which will require additional water treatment or have significant impacts on freshwater biota.

The enhanced output of nutrients in drainage waters after felling is significant, but initial analysis suggests that it will not have a major impact on long-term site fertility. The released, and subsequently fixed, phosphate, however, represents a large part of the readily available P in the system, and the availability of the fixed phosphate needs further study.

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