Impacts of afforestation on groundwater resources and quality

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Abstract Plans to double the proportion of land under forest cover in Ireland by the year 2035 have been initiated. The plan, primarily financially driven, ignores potential environmental impacts of forestry, particularly impacts on groundwater resources and quality. Since groundwater supplies almost 25% of Ireland's total potable water, these impacts are important. Field investigations indicate that afforestation leads to a reduction in runoff by as much as 20%, mainly due to interception of rainfall by forest canopies. Clearfelling has the opposite impact. Implications are that uncoordinated forestry practices can potentially exacerbate flooding. Groundwater recharge is affected by forestry, largely due to greater uptake of soil water by trees and to increased water-holding capacity of forest soils, arising from higher organic contents. Recharge rates under forests can be reduced to one tenth that under grass or heathland. Groundwater quality may be affected by enhanced acidification and nitrification under forests, due partly to scavenging of atmospheric pollutants by forest canopies, and partly to greater deposition of highly acid leaf litter. The slower recharge rates of groundwater under forests lead to significant delays in manifestation of deterioration in groundwater quality.

Résumé Des plans sont à l'étude pour doubler la proportion du couvert forestier en Irlande d'ici à 2035. Le plan, primitivement déterminé sur une base financière, ignore les impacts environnementaux potentiels de la foresterie, et particulièrement les impacts sur les ressources en eau souterraine et leur qualité. Du fait que les eaux souterraines satisfont presque 25% du total de l'eau potable de l'Irlande, ces impacts sont importants. Les études de terrain montrent que le reboisement conduit à une réduction du ruissellement d'au moins 20%, principalement à cause d'une interception de la pluie par le couvert forestier. Les coupes ont un impact contraire. Les implications sont que des pratiques forestières non coordonnées sont susceptibles d'exacerber les crues. La recharge des nappes est affectée par la foresterie, surtout à cause de prélèvements plus importants de l'eau du sol par les arbres et à cause de la capacité accrue des sols forestiers à retenir de l'eau, conduisant à de plus fortes teneurs en matières organiques. Les taux de recharge sous les forêts peuvent être réduits d'un dixième par rapport à la prairie ou à la lande. La qualité de l'eau souterraine peut être affectée par une acidification accrue et par une nitrification sous les forêts, provoquées pour une part par une fixation des polluants atmosphériques par le couvert forestier et pour une autre part par un dépôt plus important d'une litière plus fortement acide. Les taux de recharge plus lente des nappes sous les forêts conduisent à des retards importants dans la manifestation de la détérioration de la qualité de l'eau souterraine.

Resumen Se han iniciado los planes para duplicar la proporción de terrenos reforestados en Irlanda hacia el año 2035. El plan, impulsado por fines económicos, ignora los impactos potenciales medioambientales de la silvicultura, y, en particular, los impactos a los recursos de aguas subterráneas y a su calidad. Puesto que el 25 % del agua potable en Irlanda es suministrada por medio de aguas subterráneas, dichos impactos son importantes. Las investigaciones de campo indican que la reforestación lleva a una reducción de la escorrentía de hasta un 20 %, fundamentalmente por la interceptación de la lluvia en las copas de los árboles, mientras que la deforestación tiene el impacto opuesto. Las implicaciones son tales que las prácticas forestales descoordinadas pueden aumentar enormemente el riesgo de inundaciones. También la recarga a los acuíferos se ve afectada por la reforestación, debido, sobre todo, al uso del agua del suelo por los árboles y a la mayor capacidad de retención de los suelos en zonas boscosas, al disponer de más materia orgánica. Las tasas de recarga en zonas boscosas pueden verse reducidas al 10 % de las estimadas en campos de hierba o brezales. La calidad de las aguas subterráneas en zonas boscosas puede verse afectada por procesos de acidificación y nitrificación adicionales, causados por la retención de contaminantes atmosféricos en las copas de los árboles, y, en parte, por la acumulación de hojas

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enormemente ácidas en descomposición. El hecho de que la tasa de recarga sea inferior en zonas boscosas causa un retardo en la detección de fenómenos de deterioro de la calidad de las aguas subterráneas.

Keywords forestry · groundwater resources · groundwater quality · Ireland · flooding

Introduction

The proportion of land under forest cover in Ireland is 8%, smaller than for any other European country (av. 33%; Stanners and Bourdeau 1995), yet Ireland has a climate that is particularly suited to the growth of trees (Irish Government 1996). It has become Irish government policy in recent years to increase rapidly the proportion of land under forestry, and a strategic plan was formulated in 1996 to more than double the proportion of land under forest to 17% by 2035. This increase is to be achieved by planting an extra 25,000 ha annually, in addition to reafforestation, up to the year 2000, and an extra 20,000 ha annually from 2001 to 2035 (Irish Government 1998). Rapid expansion of the nation's forestry sector is necessary to change Ireland's position from a net importer of timber and other forest products to a net exporter, and it is the Irish government's intention to increase annual timber production from the current 2.2 million m^3 to at least 10 million m^3, and preferably 12–15 million m^3 (Irish Government 1996). This will have the added effect of creating forestry-related rural industries and generating employment in rural areas where demand for jobs is high.

The direction of future development of Ireland's forestry sector, as set out in the government's policy document (Irish Government 1996), makes it clear that the main impetus driving the policy is economic. A major aspect of the policy is the intention to convert agricultural land to forestry rather than to utilise some of the bleak open moorland that covers a large proportion of western Ireland for other purposes. Incentives introduced in 1980 to landowners in the western region to convert their land to forestry were largely ineffective until the level and range of economic support offered to the landowners was substantially increased (Gairdner 1993). The decision taken in the 1990s to concentrate on using agricultural land was based on the yield potential of the land; the minimum acceptable yield of timber for land suitable for conversion to forestry is yield class 18, representing a yield of 18 m^3 of timber ha^{-1} $year^{-1}$ (Irish Government 1996).

As in the 1980s, the intention is to achieve growth in forest areas primarily through promotion of private sector planting rather than by a marked increase in state forestry holdings. This will be accomplished through a series of grant schemes to farmers to convert agricultural land to forestry. Because it is EU policy to reduce farm output, the government is able to finance its own grant scheme through deployment of EU grants to induce farmers to take land out of agricultural production. The rapid conversion of thousands of hectares of predominantly agricultural land to forestry will have major environmental impacts, such as changes in landscape patterns and wildlife habitats. Potential conflicts with archaeological-heritage sites and fresh-water fisheries have also been identified. In addition, there may occur other less obvious impacts, that only become manifest some time after the forest is established, or during certain stages in the management of the plantation.

Groundwater, a major global resource, frequently suffers from the "out of sight, out of mind" syndrome. Ireland depends on groundwater for approximately 25% of total water supplies (Environmental Protection Agency 1999), but in many rural areas groundwater contributes a considerably higher proportion, and certain parts of the country depend almost totally on groundwater.

Group Water Schemes (GWS), which mostly exploit groundwater, supply about 150,000 Irish households with potable water, i.e., approximately 25% of all households in the country. Many of these GWS have received government assistance in the form of substantial grants, to supplement the significant investment of the shareholders themselves. Therefore, prudence suggests that the chosen water supplies should continue to be able to provide good quality water in appropriate quantities for the foreseeable future. A major change in land use, such as the conversion of agricultural land to forestry, could represent a potential threat to the viability of some GWS and private water supplies. As a consequence, the effectiveness of the investment of the Irish government in GWS could be reduced. Therefore, the potential impacts of land-use changes, including plantation forestry, on groundwater resources should be fully understood.

Sustainable management of natural resources is essential to ensure that future generations can continue to enjoy the same benefits as the present generation. One such benefit is access to abundant and pure fresh water. Fresh water, both surface water and groundwater, is a critical resource to mankind, which on a global scale is increasingly under threat. In contrast to surface-water sources, most groundwater requires little or no treatment before use in domestic or industrial situations, making groundwater an extremely economic option for many local authorities and industries (Gray 1994). However, contrary to popular belief, groundwater is not an infinite, renewable resource. Once contaminated or depleted, groundwater may take decades or centuries to be replenished or for the pollution to be flushed out. Thus it is especially important that the potential sources of contamination or depletion are identified and quantified.

The potential impacts of forestry on water resources are not mentioned in the strategic plan for afforestation development for Ireland. This paper presents a review of the impacts of afforestation on groundwater and to a lesser extent on surface water with which there is a significant interlinkage, and considers their implications for forestry management in Ireland.

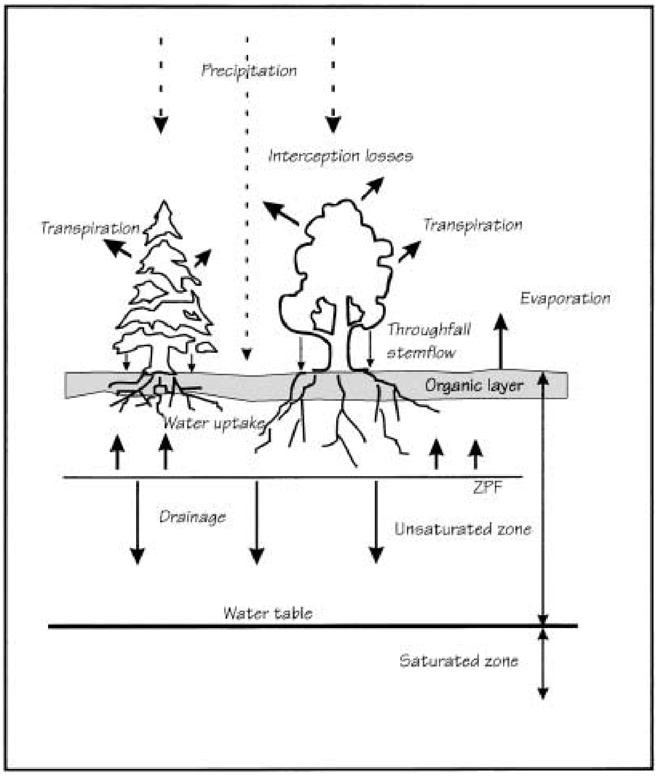

Fig. 1 Relation between forests and water balance

Impacts of Afforestation on Groundwater

A substantial body of research over nearly a century has indicated two potential impacts of afforestation on groundwater: (1) impact on the quantity of water percolating down to the water table, i.e., a potential reduction in recharge of groundwater; and (2) impact on water quality, specifically acidification (i.e., reduced pH), and changes in nitrate levels in groundwater. Of these two impacts, the first appears to be by far the most important. Most research on forestry impacts on hydrology has been concerned with surface water, because the impacts of forestry on groundwater are far more complex and difficult to measure and, therefore, to quantify. Although the processes controlling the movement of water into and through the subsurface are complex, some mathematical estimates of groundwater recharge under forests have been presented in detailed case studies. Nevertheless, it is rarely possible to make precise calculations of the rates of recharge of groundwater, and it is generally necessary to make assumptions and extrapolations in order to assess the impacts of forestry on groundwater resources. Some of the processes controlling water balance and groundwater recharge through forests are illustrated schematically in Fig. 1.

Water Budget

The influence of forests and forest clearance on water supplies and flooding has been of interest to hydrologists for almost a century. The main impact of afforestation on water resources, regardless of the type of forest cover, is a reduction in water yield (i.e., the proportion of total rainfall reaching the ground surface to undergo infiltration or surface runoff) associated with afforestation (Bosch and Hewlett 1982; McCulloch and Robinson 1993). By contrast, clearfelling, i.e., reduction in forest cover, increases water yield (Hibbert 1967). These relationships were established as early as the 1920s, as a result of two separate 20-year catchment experiments in the Bernese Emmental region of Switzerland (Engler 1919) and at Wagon Wheel Gap, Colorado, USA (Bates and Henry 1928). In the intervening 70 or so years, nearly 100 additional catchment experiments have been undertaken in different parts of the world, under various climatic conditions, particularly in relation to mean annual precipitation (MAP). All of these studies have indicated the above relationships, to a greater or lesser extent (Bosch and Hewlett 1982; McCulloch and Robinson 1993).

Reduction in water quantity by forests is due to interception of rainfall by the forest canopy (Horton 1919), leading to a loss of rainwater reaching the ground surface. Interception by coniferous forests is substantially greater than for broadleafed (Farrell et al. 1998). Interception results in a reduction in surface runoff (estimated to be in the order of 20% for upland areas in the UK) (Calder and Newson 1980) and erosion (Painter et al. 1974), and ultimately in streamflow. Both negative and positive impacts accrue from this relationship. On the positive side, flooding is partially alleviated and flood crests are reduced (Hoyt and Langbein 1955). Negative impacts include the drying up of streams and a significant overall reduction in water yield (Bosch and Hewlett 1982), which is particularly serious in situations where forestry has been established in the catchments of rivers harnessed for hydroelectric power or used for public water supplies (Calder and Newson 1980). Furthermore, reduced summer low flows (Robinson et al. 1991), attributed to higher summer transpiration and evaporation, as well as decreased groundwater recharge in winter arising from increased winter evaporation (Bates and Henry 1928), have significant consequences for agriculture.

Intercepted rainwater undergoes evaporation directly back to the atmosphere. This process is referred to as interception loss (Stewart 1968), and for coniferous forests in the UK and Ireland it is roughly twice that of adjacent grassland areas (Calder and Newson 1979). In the UK and Ireland, about half of the interception loss occurs during precipitation, whilst the rest occurs in the hours immediately following precipitation (Harding et al. 1992). Interception loss depends on the climate (primarily rainfall and wind regimes and evaporative demand during rainfall) and the structure of the vegetation, which determines its water-holding capacity and its roughness (Hall et al. 1996). Interception losses expressed as a fraction of annual precipitation for the British uplands are about 30–40%, their values decreasing with increasing rainfall (Harding et al. 1992). In view of the climatic similarities, similar interception losses could be expected for Ireland. Interception is greater for coniferous and eucalyptus forest cover than for deciduous species. A very crude estimate of the change in annual water yield per 10% change in forest cover is ~40 mm for conifers and ~25 mm for deciduous species (Bosch and Hewlett 1982), but these are global figures that do not take into account local climatic conditions, particularly MAP.

Forests also affect the water budget by transpiration loss, the transfer of water from the soil to the atmosphere through the tree roots, trunk, and leaves. In the UK, the water use of coniferous plantations ranges from about 600 mm/year in the east, where transpiration loss is the main component, to about 1,200 mm/year in the upland areas of the west, where interception loss dominates (Hall et al. 1996). Fewer data are available on the water use of deciduous trees. However, a study by Harding et al. (1992) on two sites in southern England indicates that annual transpiration losses on clay and chalk bedrock are 327 and 372 mm respectively for ash plantations. Similar but slightly lower values are recorded for beech. This finding is comparable with the overall value of ~330±35 mm presented by Roberts (1983) for the annual transpiration loss from various forest sites in northwestern Europe. Transpiration totals are remarkably similar for both coniferous and deciduous forests, indicating a strong physiological control on transpiration. Surprisingly, in their study, Harding et al. (1992) determined that the total measured water use for the beech and ash plantations on the clay and chalk bedrock (380 mm), representing the combined interception and transpiration loss from the forest plantations, is lower than the value for grassland evapotranspiration (430 mm) estimated by a MORECS (Meteorological Office Rainfall and Evaporation Calculation System) calculation (Thompson et al. 1981). However, the forest measurements were made in a year of very low summer rainfall (1989), and the potential evapotranspiration total for the forest sites should normally be in the order of 700 mm. Furthermore, based on the value for reduction in water yield due to interception loss of 10 mm/year per 10% grassland cover (Bosch and Hewlett 1982), i.e., 100 mm/year for total grassland cover, the transpiration loss for grassland is therefore about 330 mm/year, very similar to that for trees.

In Ireland, meteorological records show that precipitation is relatively evenly distributed throughout the year and normally greatly exceeds evapotranspiration for most of the year. Only for a month or two in the summer may this situation be reversed, more commonly in the drier east than in the west, and the water deficit is rarely great. Due to mild average temperatures in the winter months, the growth of trees continues throughout most of the year, and water limitation affects growth only for very short periods during relatively dry summer spells.

The energy used in evapotranspiration (both transpiration and interception) from vegetation was considered until recently to come solely from the input of solar radiation, and it was assumed that the energy used up in evaporating intercepted water from forests could not also be used for transpiration (Penman 1948, 1963). The result would have been a reduction in transpiration. However, recent research has shown that the extra energy demands by forests for evapotranspiration can come from advection, i.e., the cooling of the air mass above and within the forest (Calder and Newson 1980). Forests are better able to make use of this advective energy than shorter crops because they generate a more turbulent wind regime above their canopies. In wet conditions during and immediately after rainfall, exploitation of this energy source by forests is so efficient that rates of evaporation of intercepted water can be as much as 10 times those from grass (Calder and Newson 1980). This is the principal cause of the extra water losses by evaporation from forests in upland areas compared with grasslands.

The most significant effect of forest cover on the recharge of groundwater is that rainfall infiltration through the forest canopy and the underlying forest litter into the soil is more effective than through grasslands or agricultural crops (McCulloch and Robinson 1993). This would suggest that recharge should be enhanced by forest cover. However, the rooting depths of trees are generally greater than for short crops, and the water requirements of trees are significantly greater than those of grass or crops. The combined result is a greater removal of water from the soil by tree roots and to deeper levels in the soil profile, which outweigh the effect of higher infiltration (McCulloch and Robinson 1993). Calder et al. (1992) observed that the annual water use of trees without access to groundwater can be as much as twice that of an annual agricultural crop. This reduces the moisture content of the soil, leading to a decreased proportion of the soil water in the unsaturated zone percolating (draining) down to the water table. This effect is confirmed by a study of a coniferous forest in East Anglia (Cooper 1980), which showed that drainage under the forest is 44% less than under grass in a clearing. The increased water-holding capacity of forest soils is due mainly to the higher organic content of the soils (Virzo De Santo 1992). A further study in the West Midlands of England (Moss and Edmunds 1989) revealed that the rate of recharge of the water table under forest is at least three times lower than under heathland (~330 mm/year) and may be as little as about 20 mm/year. The reduced rate of recharge means that the residence time for water in the unsaturated zone under forests is significantly greater than that under heathlands, and may be as much as a decade or two (Moss and Edmunds 1992). Thus it could be many years before any change in groundwater quality that might be associated with a change in land use to forestry is reflected in the quality of groundwater pumped from adjacent water wells (Hall et al. 1996).

During dry periods (mainly summer), removal of soil water by direct surface evaporation and by plant roots gives rise to a soil-moisture deficit, which leads to the establishment of a zero flux plane (ZFP), dividing upward capillary-moisture movement in the upper part of the profile from downward movement (Richards et al. 1956). The soil-moisture deficit is significantly greater under forests than grassland, with the result that the depth of the ZFP, which varies with the magnitude of the soil-moisture deficit, is generally greater under trees than under grass (Cooper 1980). Where the water table is relatively close to the surface, the ZFP may intersect the water table, and groundwater from below the water table may move upward by capillary action to higher levels in the soil profile in order to reduce the soil-moisture deficit. Ultimately, such groundwater may be lost by a combination of plant removal and evaporation leading to a lowering of the level of the water table (a negative recharge). Thus, despite enhanced infiltration of rainwater into the soil under forest cover, a reduction in recharge as a proportion of total precipitation probably occurs, and during summer dry spells a negative recharge (discharge) may even occur.

One further impact of forestry on groundwater resources, is the overall balance between groundwater recharge and discharge in relation to streamflow. In humid areas during dry spells when stream levels are low (low flow conditions), the water level of the streams corresponds to local water table levels, and stream levels are maintained by groundwater recharge (Meybeck et al. 1996, Fig. 6.2). During and immediately after heavy and prolonged rainfall, streams rise rapidly towards their flood crests, whereas groundwater recharge is delayed, so stream levels temporarily rise above the adjacent water table level, leading to a reversal of recharge/discharge relations. Thus streams recharge groundwater via discharge through their banks, and the water table accordingly rises (Meybeck et al. 1996, Fig. 6.2). The rate of discharge from the stream into the ground depends on the hydraulic head between the stream level and the water table, and the hydraulic conductivity of the stream bank deposits. As the level of the water table adjacent to the stream rises due to recharge, and the stream level gradually recedes after rainfall ceases, rate of recharge from the stream decreases. Eventually the stream level drops below the level of the water table, bringing about another reverse in recharge/discharge relations, so that groundwater once more recharges streams.

The extent of recharge of the water table by streams depends on the amount of precipitation and duration of any rainfall event, the level to which the stream rises, and the length of time that the level of the stream exceeds the water table level. Any factor that reduces the flood crests of streams will alter the extent of recharge of groundwater by streams during high flow episodes. Thus, afforestation, in reducing the amount and proportion of runoff (Calder and Newson 1980), and diminishing flood crests (Hoyt and Langbein 1955), leads to a decrease in the extent of recharge of groundwater by streams during storm episodes. The net result would be a further reduction in groundwater recharge, over and above that due to decrease in percolation under forests, leading to an overall lowering of the water table over the long term. Strong support for this argument is the evidence of streams drying up after forest plantations have been established (Bosch and Hewlett 1982), reflecting an overall lowering of the water table to a level below that of the stream bed.

In conclusion, it is realised that the above is an extremely generalised and necessarily simplified discussion of the impact of forestry on the overall balance between groundwater recharge and discharge in relation to streamflow. Also, to define the impact in detail at specific sites, one must also consider the complex interrelationships among numerous additional factors such as land cover, rainfall, infiltration, evapotranspiration, the spatial distribution of water-table and piezometric altitudes, and the anisotropic nature of the surface-water/groundwater circulation system. The overall impacts of afforestation on the water budget, however, appear to be a reduction in water yield, a change in the proportions of runoff, infiltration, and evapotranspiration, and an overall decrease in the recharge of groundwater, leading to a lowering of local water tables.

Water Quality

The quality of surface and groundwaters is a function of a various factors (Hall et al. 1996):

- The chemistry of the rainfall and other atmospheric inputs.
- The weathering of minerals in soils and rocks.
- Inputs derived from diffuse agricultural sources (fertilisers or enhanced mineralisation).
- Man-made inputs from point sources such as individual factories or more generally from urban areas.
- The amount of rainfall and evaporation.

Surface water is generally more susceptible to a deterioration in water quality than groundwaters. Soil and geological formations can mitigate groundwater contamination in the unsaturated zone before pollutants reach the water table. Even in the saturated zone, exchange processes and reactions between solid mineral grains and the groundwater can bring about attenuation of contaminants in the groundwater. The two most critical aspects of water quality related to forest cover are *acidification* and *nitrification*, both of which have received considerable attention over the last two decades. Some of the processes that control them are illustrated in Fig. 2.

Acidification

Trees scavenge atmospheric pollutants about two to three times more efficiently than short crops (Brechtel 1992) because of their greater aerodynamic roughness and larger leaf area (Harding et al. 1992). Because conifers are aerodynamically rougher, they are likely to be more efficient scavengers than broadleaf species. The role of forests in acidifying upland streams is now primarily attributed to the enhanced scavenging of acid pollutants, i.e., deposition on the tree canopy of industrial emissions of acidic oxides, principally SO_2 and NO_x derived from fossil-fuel combustion, non-ferrous metal smelting, fertiliser and sulphuric acid manufacture, and other industrial

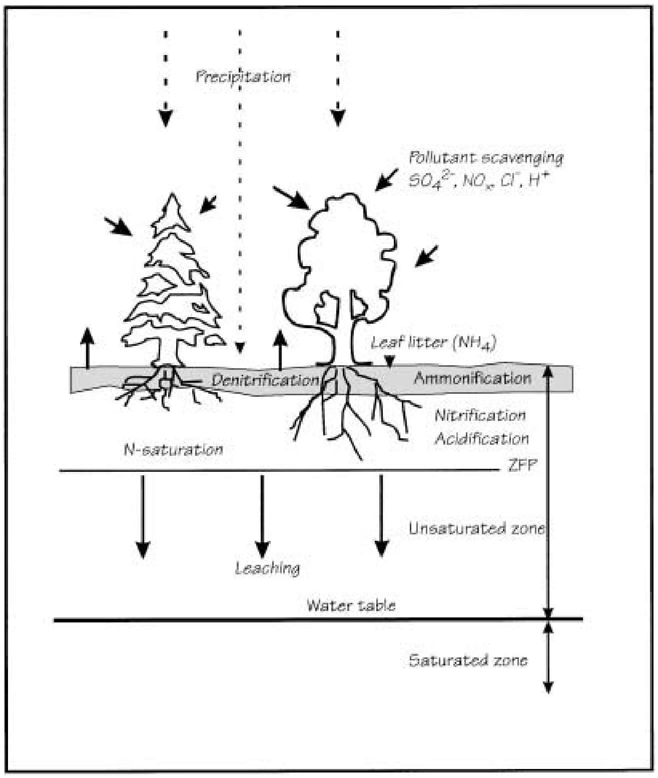

Fig. 2 Relation between forests and water quality

processes, as well as ammonia emissions from agricultural sources (Meybeck et al. 1989). Scavenging results both by wet deposition of cloud and mist droplets, which contain significantly higher ionic concentrations than larger rain droplets, and by dry deposition through impaction of particles and from reactive gases (Fowler et al. 1989). These pollutants are then washed down to the ground surface by stemflow and leafdrip processes, where they either infiltrate into the ground or are washed into surface streams. In the UK uplands, marked reductions in salmonid fish stocks, stream invertebrate diversity, and wildlife that are dependent on rivers for their food supply (United Kingdom Acid Waters Review Group 1988) are attributed to acidification of upland streams.

Additional effects associated with forest development, such as loss of base cations from the soil to the biomass, increased supply of highly acidic organic matter (leaf litter) to the soil, and changes in hydrological flow pathways (Rosenqvist 1980; Krug and Frink 1983; United Kingdom Acid Waters Review Group 1988; Jenkins et al. 1990) also result in acidification. The loss of base cations from the soil and the increased supply of acidic leaf litter strongly influence the acidification of water infiltrating into the ground and percolating down through the unsaturated zone, whilst the marked reduction in runoff ensures that most of the atmospheric pollutants scavenged by the forest canopy are carried downward by infiltrating rainwater into the forest soil. Increased concentrations of H^+, SO_4^{2-}, and NO_3^-, scavenged from the atmosphere and the forest litter and humus of the surface layers, give rise to highly acidic infiltrating water (pH<5). This water interacts with the soil or bedrock as it percolates downwards, leading to gradual loss of base cations (principally Ca^{2+} and Mg^{2+}) from these media. Highly siliceous bedrock and overburden are especially sensitive to acidification (Meybeck et al. 1989).

The interaction between groundwater and the media through which it passes leads to weathering and ion-exchange processes in the soil and bedrock, reducing groundwater acidity. The magnitude of this neutralising effect is dependent on the mineral composition of the rock or soil, particularly the carbonate and clay contents and the quantity of organic matter present, as well as on the residence time of water passing through. However, dissolution of other minerals, e.g., aluminosilicate minerals such as feldspar, may, in the absence of carbonates or high clay contents, also have a significant neutralising effect (Dahmke et al. 1986; Moss and Edmunds 1992). Carbonate bedrock, bedrock with a carbonate cement or overburden with abundant carbonate float (such as might be deposited by a glacier that had sampled a carbonate terrain) have the greatest buffering capacity and would normally completely neutralise highly acid groundwaters. A range of rock types, including intermediate to mafic volcanic and plutonic igneous, mafic, pelitic, and calcsilicate metamorphic rocks, and argillaceous sediments, have a moderate buffering capacity, whilst acid igneous rocks, quartzofeldspathic metamorphic rocks, and arenaceous sediments have a low buffering capacity. Clay-rich overburden could be expected to have a moderate to high buffering capacity, depending on the clay-mineral types present, whilst peat accentuates acidification. In Ireland much of the central and eastern parts of the country are blanketed by thick clay-rich drift deposits with moderate to high buffering capacity, whereas in the west and on higher ground, rocks with low buffering capacities are either exposed or overlain by variable thicknesses of peat.

Organic processes that affect groundwater composition in the unsaturated zone usually increase the acidity of the percolating groundwater, and include nitrification, base-cation uptake by vegetation, production of organic acids in decaying vegetation, and oxidation of reduced forms of sulphur (Reuss et al. 1987). Organic-rich soils underlying forests buffer interstitial water at a lower pH than heathland soils (Moss and Edmunds 1992), which in turn are more acid than grassland soils. Furthermore, the rate of assimilation of strong acid anions, e.g., SO_4^{2-}, is higher in forest soils.

Inorganic processes of groundwater mediation include dissolution of carbonate by acidified groundwater, to produce Ca^{2+} and less frequently Mg^{2+}; rapid cation-exchange reactions, principally with clay minerals; and slow mineral weathering. In carbonate-free rocks and soils, the buffering capacity is dependent on the rate of mineral weathering reactions in relation to water flux, and the magnitude of the acid-neutralising capacity of the mineral reactions (Moss and Edmunds 1992). Rates of mineral reactions are generally dependent on the level of microbial activity and on the pH of interstitial water. Greater microbial activity, associated with the higher organic content of forest soils, and the presence of such microorganisms as mycorrhizal fungi enhance the rate of mineral weathering, as does the lower pH values of groundwater under forests. Dissolution rates of common rock-forming minerals, such as alkali feldspar, are relatively enhanced under such conditions. However, rates of mineral weathering above the water table are also critically dependent on the flux of water draining downward, which in forest soils is significantly lower than in grassland soils. Thus overall, mineral dissolution in forest soils is likely to be somewhat slower than in grassland soils.

Weathering of parent aluminosilicate minerals, e.g., alkali feldspar, produces a variety of secondary clay minerals, such as kaolinite, montmorillonite, and gibbsite (Meybeck et al. 1989). Where groundwater has a $pH<4$, gibbsite and other aluminium hydroxides may undergo further dissolution, releasing toxic labile monomeric Al^{3+} (Driscoll 1985), which can become the main proton sink (Kreutzer et al. 1998). In general, Al saturation exists only at the lower boundary of the main root zone, but in maritime environments, where forests are subject to seasalt deposition, the ionic field strength effect of seasalt input can lead to deep-reaching Al saturation and acidification of soil water (Kreutzer et al. 1998).

Increasing acidity may also mobilise other trace metals, e.g., Cd, Mn, Fe, As, and Hg from soils (Meybeck et al. 1989). Mobile anions, e.g., SO_4^{2-}, arising from atmospheric deposition, may leach positively charged ions to maintain electroneutrality (Seip 1980). In well-buffered unsaturated zones, SO_4^{2-} is leached down to the saturated zone accompanied by the base cations Ca^{2+} and Mg^{2+}, whereas in poorly buffered unsaturated zones, H^+ and Al^{3+} are leached, resulting ultimately in decreases in alkalinity and pH (Meybeck et al. 1989). As indicated above, forest soils may be able to absorb a large proportion of the deposited SO_4^{2-}, which may temporarily inhibit groundwater acidification. Nitrate appears to have little role in acidification of groundwater, except where forest soils may become saturated with nitrate, under which circumstances NO_3^- may contribute significantly to acidification of groundwater (Henriksen and Brakke 1988). This latter effect can be enhanced under certain climatic conditions, such as drought followed by rewetting, provided air temperatures are high enough, circumstances that generally do not pertain under current Irish climatic conditions but which could become more prevalent in a 'global warming' scenario (Ryan et al. 1998).

Nitrification

Nitrogen may be scavenged by forests as NO_x, mainly derived from fossil-fuel combustion, fertiliser manufacture, and other industrial processes, and from NH_4 emissions from agricultural sources, such as animal manures. Nitrogen in groundwater may be derived from soil organic matter and fertiliser application (McLaughlin et al. 1985; Hall et al. 1996; Zhang et al. 1998); the latter is the dominant source of groundwater nitrate contamination (van der Voet et al. 1996). Both the increased efficiency of pollutant scavenging by forests and the significantly enhanced supply of organic matter to the soil can lead to 'nitrogen saturation' of forest soils (Stevens and Hornung 1988; Gundersen and Rasmussen 1990), i.e., nitrogen in excess of the demands of tree growth, particularly in mature and slowly growing forests (Hall et al. 1996). Once a forest becomes N-saturated, the excess nitrate is leached by percolating groundwater down to the saturated zone (Johnson 1992).

Nitrogen is a fundamental nutrient for plant growth, and an inadequate supply is growth limiting (Verhoef et al. 1992). Physiological processes involving nitrogen in the soil are complex and include (Meybeck et al. 1989):

- Nitrogen fixation – the reduction of nitrogen gas (N_2) to ammonia (NH_4) and organic nitrogen by microorganisms.
- Assimilation of inorganic forms, ammonia, and nitrate (NO_3^-) by plants and microorganisms to form organic nitrogen, e.g., amino acids.
- Heterotrophic conversion of organic nitrogen from one organism to another.
- Ammonification of organic nitrogen to produce ammonia during the decomposition of organic matter.
- Nitrification, i.e., oxidation of nitrogen and ammonia to nitrate and nitrite (NO_2^-).
- Denitrification, i.e., reduction of nitrate and nitrite to nitrous oxide (N_2O) and nitrogen gas.

The major soluble forms of nitrogen in water are nitrate and ammonia. High levels of organic matter in forest soils promote ammonification, provided that decomposition of the organic matter keeps pace with its accumulation. In the event that it does not, such as in young or fast-growing forests (Nihlgård 1997), excess nitrogen is converted to nitrate. Nitrification gives rise to a net acidification effect if the nitrification rate in the soil (an H^+-producing process) exceeds the rate of uptake of nitrate by roots and soil microorganisms (an H^+-consuming process) (Bredemeier and Marmann 1992). Thus the nitrate load of seepage water depends on the N status of the ecosystem, rather than N deposition above a deposition threshold, which for coniferous forests in some northwestern European countries is 20 kg N ha^{-1} $year^{-1}$ (Kreutzer et al. 1998).

Soil acidification leads to the loss of essential nutrients, such as the base cations Ca and Mg, which in turn may lead to forest decline (Becquer et al. 1992). Leaching of nitrogen from forest soils usually takes place after nitrification (Farrell et al. 1992), coinciding with increased acidification and accumulation of nitrate and toxic aluminium concentrations in the groundwater (Tietema et al. 1992).

Nitrification can be reversed, and thereby acidification of the soil inhibited, by denitrification, an anaerobic respiratory process, which results in the return of nitrogen, principally in the form of nitrous oxide, a greenhouse gas, to the atmosphere (Virzo De Santo 1992). Which of these competing physiological processes predominates in forest soils is dependent on a variety of factors, including soil composition and texture, soil-moisture content, soil-water pH, quantity of organic matter in the soil surface layers, climate (particularly rainfall and temperature), microbial population, and availability of carbon and oxygen (Virzo De Santo 1992). Seasonal variation in climatic conditions and the growth activity of trees can lead to switches from nitrification to denitrification and vice versa (Alfani et al. 1983), as can short-term weather variations such as rainfall (Binstock 1984). In addition, clearfelling can affect the balance of nitrification/denitrification processes, because this leads to a two- to three-fold increase in N-mineralisation (Granhall 1992; Khanna et al. 1992). At one free-draining site in a stand of Norway spruce in Ireland, the concentration of nitrate in soil water increased even more dramatically after clearfelling. In deep soil water at a depth of 50 cm, the mean nitrate concentration in the year subsequent to clearfelling increased by a factor in excess of 20 relative to the mean nitrate concentration over the 6-year period prior to clearfelling. For a significant portion of this period, nitrate values exceeded maximum concentrations permissible under EU directives for drinking water (Jones et al. 1998). Where slash is left on the ground after clearfelling, nitrification is enhanced significantly, whereas removal of surface organic debris during site preparation for replanting favours denitrification processes (Virzo De Santo 1992). Fertiliser addition and liming to improve soils and promote tree growth also promote N-mineralisation and nitrification (Khanna et al. 1992), but the latter, by increasing the base-cation supply to the soil, also reduces acidification (Rapp 1992). Fires may also affect the nitrogen balance in forest soils temporarily, bringing about a two- to five-fold increase in soil-mineral N levels (N-mineralisation) as a consequence of heating in the surface (0–5 cm) soil (Keith and Raison 1992). This increase could also lead to enhanced leaching of nitrate by groundwater.

Finally, human disturbances overall, whether in the form of atmospheric nitrogen deposition arising from agricultural or industrial activities, or exploitation of forests as a resource, give rise to nutrient imbalances, leading to aggravated acidification of groundwater (Nihlgård 1997). Forest-management practices, such as fertiliser application and harvesting of forests by clearfelling, bring about an enhanced supply of nitrogen, which tends to result in more rapid tree growth, leading to greater demand for nutrients. This then becomes growth limiting, so nitrogen saturation results and leads to addition of strong nitric acid to the soil water, thus lowering its pH level to between 4 and 4.4 (Nihlgård 1997). This in turn gives rise to increased leaching of both nitrates and base cations, intensified acidification, and buildup of nitrate in the groundwater.

Thus the main impacts of ‘nitrogen saturation’ of forest soils are: (1) increased nitrification, leading to acidification and leaching of nitrate by percolating groundwater into the saturated zone; and (2) enhanced denitrification, with loss of nitrogen from biological systems to the atmosphere, mainly in the form of the greenhouse gas nitrous oxide.

Discussion

The above review indicates that land use in general, and forestry in particular, has major impacts on both water resources and water quality. These impacts are more easily documented for surface water, but are just as important with respect to groundwater.

Water Resources

Ireland is not usually subject to serious drought problems, and water shortages are generally confined to the drier, eastern part of the country. Localised water shortages are usually short-lived and rarely severe. Ireland is more subject to flooding than drought, particularly in the wetter, western parts of the country, where thousands of acres of low-lying agricultural land are commonly inundated for long periods during the winter months. Forestry research (e.g., Hoyt and Langbein 1955) has indicated that a strong connection exists between forest cover and flooding in individual catchments. Strategic afforestation of higher ground could directly, by reduction of the quantity of rainwater available for direct runoff through significantly increased interception, and indirectly, by enhanced infiltration and the slower rate of recharge of groundwater under forests, alleviate flooding in the flood-prone areas of the west. However, the policy of using better quality agricultural land for new forestry plantations, as outlined in the government’s strategic plan for the forestry sector, suggests that most of the land that would be planted would be in the central and eastern parts of the country where, rather than flooding, problems of water shortages could be exacerbated by afforestation. The cost of flood damage to the nation each year is considerable, and the economic benefits of flood alleviation probably exceed the economic advantages of conversion to forestry only of land with higher yield potential. No cost–benefit analysis comparing the government policy for afforestation with economic benefits of flood alleviation was undertaken in the strategic plan.

The impact of forestry on flooding also has major implications for forest management. Not only can afforestation alleviate flooding in flood-prone areas, by controlling runoff, but also harvesting of forests can lead to flooding, by the removal of the controls on runoff, leading to exposure of bare ground to erosion, and an associated increase in siltation of rivers. The link between forestry and flooding presents the potential, through careful planning and management of forestry plantations, to control to a significant extent flooding in Ireland and its associated hardship and damage. For example, in relation to flooding, the proportion of any one catchment to be planted within a particular timeframe and, equally importantly, the proportion of forestry within a catchment to be harvested over any period of years, are critical. The most effective method of establishing such a coordinated national management strategy would be through strong centralised control of the forestry industry, with economic objectives balanced with the overall public interest.

No definitive direct linkage has yet been established between afforestation and groundwater depletion. This is mainly due to the difficulty in planning and executing field experiments of sufficiently long time-span, ranging from a period of pre-forestry through the conversion of land to forestry, the growth to maturity of the forest, and its subsequent harvesting and return of the land to an unforested state. However, the evidence for a reduction in the rate of recharge beneath forest cover suggests that a lowering of the water table would be the long-term effect of a change in land use to forestry. A number of long-term groundwater investigations at carefully monitored sites need to be undertaken in order to examine the impact of forests on groundwater resources, particularly in Ireland, where a significant proportion of the population is dependent on groundwater as a source of potable water.

Estimation of the reduction in the proportion of rainwater reaching surface streams as a result of a change in land use to forestry is relatively uncomplicated. However, calculation of groundwater recharge rates for any given situation is untenable in the absence of detailed information on the nature and thickness of overburden, primarily the proportion of clay minerals present, the specific clay-mineral content, the organic content, and the moisture potential, particularly of the surface soil layers. Also essential is a measure of the thickness of the unsaturated zone, and whether the mode of movement of percolating water through bedrock within the unsaturated zone is primarily by intergranular flow or by fissure flow.

Water Quality

Assuming normal vertical recharge to an unconfined aquifer by intergranular flow, the rate of recharge below forest cover may be as little as 20 mm/year, and the residence time for groundwater in the unsaturated zone may be a decade or more, both depending on the spatial distribution of hydraulic conductivity of the zone. Any deterioration in the quality of recharge water due to afforestation could take at least a decade to reach the water table and become manifested in public-supply wells.

Research indicates that trees are highly efficient scavengers of pollutants, and several studies have shown that the scavenging may lead to acidification and nitrification of groundwater (Fowler et al. 1989; Hauhs et al. 1989; Meybeck et al. 1989; Gundersen and Rasmussen 1990; Tietema et al. 1992). This effect is most critical in regions leeward of highly industrialised areas. In Europe, high levels of scavenging of atmospheric pollutants, nitrogen saturation, and acidification have been identified in central European forests. The UK and some western and northern European countries display significantly lower mean values of acid deposition (2–4 times less) from atmospheric pollution (i.e., NH_4-N, NO_3-N, SO_4-S and Cl) in open-field sites than central European countries (Brechtel 1992). However, in view of the prevailing southwesterly direction of winds affecting Ireland, and the limited industrial development particularly in the western parts of the country, Ireland is probably subject to a much lower atmospheric pollution risk than almost anywhere else in Europe. Nonetheless, even in Ireland anthropogenic deposition of N and S is two to three times that of pre-industrial deposition (Kreutzer et al. 1998). Furthermore, Irish forests, both coniferous and broadleaf, are subject to considerable seasalt deposition (Farrell et al. 1998), which can give rise to Mg^{2+} in forest soils in coastal regions far in excess of the demand of trees, and also can induce acidification of groundwater by control of Al solubility (Kreutzer et al. 1998).

Forestry can, as indicated above, give rise to acidification and nitrification of groundwater through loss of base cations from the soil to the biomass, together with a significantly enhanced organic component in the soil surface layers due to the increased supply of highly acidic leaf litter. Fertiliser application during the early stages of plantation development to promote tree growth could also have the negative effect of promoting acidification and nitrification of groundwater, as can harvesting methods. Furthermore, excess nitrogen loads, due to poor farming practices, may be inherited in land converted from agriculture. There exists a clear need to monitor the different impacts on groundwater quality of coniferous, broadleaf, and mixed forests under different geological and hydrological conditions over a long timescale, in order to establish trends and identify controls.

Conclusions

Major land-use changes inevitably give rise to environmental impacts, some positive and some negative. It is essential that environmental effects are thoroughly investigated in advance, so that potential negative consequences are identified and balanced against positive effects and economic advantages. Where possible, measures to alleviate negative impacts should be instituted.

There is little question that restoration of forests to substantial areas of Ireland would have many positive environmental impacts, in addition to the obvious economic ones. However, in promoting the advantages of this scheme, the negative impacts should not be hidden away or indeed lost sight of. The economic merits of the Irish government's plan to rapidly expand the forestry sector are unquestionable, but some of the environmental implications of this policy have not been identified, and the potential costs in terms of resource conflicts and adverse environmental impacts have not been properly evaluated in the economic equation. Furthermore, some

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potentially damaging environmental impacts may have been ignored.

One of the 'hidden' environmental conflicts, which the strategic plan fails to address, is the impact of forestry on water resources and water quality, both surface water and groundwater. In view of the dependence on groundwater of a significant proportion of the Irish population, this is a major oversight. Abundant research evidence exists on the impacts of forestry on surface water, but comparatively little exists on groundwater, because of the inherent difficulties in measuring and thereby quantifying impacts. Thus, long-term research programmes are urgently needed to investigate, in particular, recharge of groundwater under varying conditions, and also the effects of acidification and nitrification processes in forest soils of differing forest types and underlying hydrogeology. These studies are necessary to protect groundwater as a natural resource, and to help ensure that the development and exploitation of one natural resource does not have adverse impacts on another extremely important resource.

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References

Alfani A, Fioretto A, Virzo De Santo A, Russo G (1983) Denitrification potential of beech soils as influenced by the seasonal cycle. Pedobiologia 25:149–156

Bates CG, Henry AJ (1928) Forest and streamflow experiment at Wagon Wheel Gap, Colorado. Mon Weather Rev Suppl 30:1–79

Becquer T, Herbillon A, Merlet D, Boudot JP, Rouiller J (1992) Importance of the nitrogen and sulfur cycles in the proton budget in a declining fir forest and its relation to aluminium toxicity. In: Teller A, Mathy P, Jeffers JNR (eds) Responses of forest ecosystems to environmental changes. Elsevier, New York, pp 679–680

Binstock DA (1984) Potential denitrification in an acid forest soil; dependence on wetting and drying. Soil Biol Biochem 16: 287–288

Bosch JM, Hewlett JD (1982) A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. J Hydrol 55:3–23

Brechtel HM (1992) Impact of acid deposition caused by air pollution in central Europe. In: Teller A, Mathy P, Jeffers JNR (eds) Responses of forest ecosystems to environmental changes. Elsevier, New York, pp 594–595

Bredemeier M, Marmann P (1992) Acidification pulses due to nitrification in forest soils. In: Teller A, Mathy P, Jeffers JNR (eds) Responses of forest ecosystems to environmental changes. Elsevier, New York, pp 732–734

Calder IR, Newson MD (1979) Land use and upland water resources in Britain – a strategic look. Water Resour Bull 16: 1628–1639

Calder IR, Newson MD (1980) The effects of afforestation on water resources in Scotland. In: Land Assessment in Scotland, Proc Symp Roy Geogr Soc Edinburgh, pp 51–62

Calder IR, Swaminath MH, Kariyappa GS, Srinivasala NV, Srinivasala Murthy KV, Mumtaz J (1992) Deuterium tracing for the estimation of transpiration from trees. J Hydrol 130: 37–47

Cooper JD (1980) Measurement of moisture fluxes in unsaturated soil in Thetford Forest. Rep 66, Institute of Hydrology, Wallingford, UK

Dahmke A, Matthes G, Pekdeger A, Schenk D, Schulz HD (1986) Near surface geochemical processes in Quaternary sediments. J Geol Soc London 143:667–672

Driscoll CT (1985) Aluminium in acidic surface waters: chemistry, transport and effects. Environ Health Perspec 63:93–104

Engler A (1919) Untersuchungen über den einfluss des waldes auf den stand der gewasser (Investigations into the impact of forests on the level of surface water). Mitt Schweiz Anst Forst Versuchswes 12:636 pp

Environmental Protection Agency (1999) Water quality in Ireland 1995–1997. US EPA, Wexford

Farrell EP, Smillie GW, Collins JF, Hennessy C, McCarthy R (1992) Precipitation, throughfall and soil water chemistry in a spruce forest in C. Cork, Ireland. Ballyhooly project. In: Teller A, Mathy P, Jeffers JNR (eds) Responses of forest ecosystems to environmental changes. Elsevier, New York, pp 726–727

Farrell EP, Van Den Beuken R, Boyle GM, Cummins T, Aherne J (1998) Interception of seasalt by coniferous and broadleafed woodland in a maritime environment in western Ireland. Chemosphere 36:985–987

Fowler D, Cape JN, Unsworth MH (1989) Deposition of atmospheric pollutants in forests. Phil Trans Roy Soc London 324: 247–265

Gairdner G (1993) Incentives for private forestry – the case of the Republic of Ireland. Environ Conserv 20:50–56

Granhall U (1992) Clear-cutting of a Scots pine forest – effects on soil biology. In: Teller A, Mathy P, Jeffers JNR (eds) Responses of forest ecosystems to environmental changes. Elsevier, New York, pp 797–799

Gray NF (1994) Drinking water quality. Problems and solutions. Wiley, Chichester, UK, 315 pp

Gundersen P, Rasmussen L (1990) Nitrification in forest soils: effects from nitrogen deposition on soil acidification and aluminium release. Rev Environ Contam Toxicol 113:1–45

Hall RL, Allen SJ, Rosier PTW, Smith DM, Hodnett MG, Roberts JM, Hopkins R, Davies HN, Kinniburgh DG, Gooddy DC (1996) Hydrological effects of short rotation energy coppice. ETSU B/W5/00275/Rep.:204 pp

Harding RJ, Neal C, Whitehead PG (1992) Hydrological effects of plantation forestry in north-western Europe. In: Teller A, Mathy P, Jeffers JNR (eds) Responses of forest ecosystems to environmental changes. Elsevier, New York, pp 445–455

Hauhs M, Rost-Siebert K, Raben G (1989) Summary of European data. In: Malanchuk J, Nilsson J (eds) The role of nitrogen in the acidification of soils and surface waters. Nordic Council of Ministers, Copenhagen, pp 5–37

Henriksen A, Brakke DF (1988) Increasing contributions of nitrogen to the acidity of surface waters in Norway. Water Air Soil Pollut 14:183–201

Hibbert AR (1967) Forest treatment effects on water yield. In: Sopper WE, Lull HW (eds) Forest hydrology. Pergamon, Oxford, UK, pp 527–543

Horton RE (1919) Mon Weather Rev 47:603–623

Hoyt WG, Langbein WB (1955) Floods. Princeton University Press, Princeton, NJ, USA, 469 pp

Irish Government (1996) Growing for the future: a strategic plan for the development of the forestry sector in Ireland. Department of Agriculture, Food and Fisheries, Irish Government Stationary Office, Dublin, 98 pp

Irish Government (1998) Afforestation grant and premium schemes. Forest Service, Department of Marine and Natural Resources, Irish Government Stationary Office, Dublin, 27 pp

DOI 10.1007/s100400100148

Jenkins A, Cosby BJ, Miller JD, Ferrier RC, Walker TAB (1990) Modelling stream acidification in afforested catchments: long term reconstruction at two sites in central Scotland. J Hydrol 120:163–181

Johnson DW (1992) Nitrogen retention in forest soils. J Environ Qual 21:1–12

Jones SM, Cummins T, Boyle GM, Aherne J, Farrell EP (1998) A pilot study into the effects of clearfelling on nutrient losses and sustainability. Monstac Project Final Report, Forest Ecosystem Research Group Rep 23, University College, Dublin, 62 pp

Keith H, Raison RJ (1992) Effects of prescribed fire on nitrogen cycling and tree growth in an Australian eucalypt forest. In: Teller A, Mathy P, Jeffers JNR (eds) Responses of forest ecosystems to environmental changes. Elsevier, New York, pp 889–891

Khanna PK, Meiwes KJ, Bauhus J (1992) Nitrogen dynamics in some beech forests as measure of resilience to changes due to management practices and anthropogenic impacts. In: Teller A, Mathy P, Jeffers JNR (eds) Responses of forest ecosystems to environmental changes. Elsevier, New York, pp 664–665

Kreutzer K, Beier C, Bredemeier M, Blanck K, Cummins T, Farrell EP, Lammerdorf N, Rasmussen L, Rothe A, deVisser PHB, Weis W, Weiss T, Xu YJ (1998) Atmospheric deposition and soil acidification in five coniferous forest ecosystems: a comparison of the control plots of the EXMAN sites. Forest Ecol Manage 101:125–142

Krug E, Frink CR (1983) Acid rain on acid soil: a new perspective. Science 221:520–525

McCulloch JSG, Robinson M (1993) History of forest hydrology. J Hydrol 150:189–216

McLaughlin RA, Pope PE, Hansen EA (1985) Nitrogen fertilisation and groundcover in a hybrid poplar plantation. J Environ Qual 14:241–245

Meybeck M, Chapman DV, Helmer R (eds) (1989) Global freshwater quality – a first assessment. Nitrates in freshwater. Blackwell, Oxford, UK, pp 195–217

Meybeck M, Friedrich G, Thomas R, Chapman DV (1996) Rivers. In: Chapman DV (ed) Water quality assessments, 2nd edn. E & FN Spon, pp 243–318

Moss PD, Edmunds WM (1989) Interstitial water–rock interaction in the unsaturated zone of a Permo-Triassic sandstone aquifer. In: Miles (ed) Water–rock interactions. AA Balkema, Rotterdam, Netherlands

Moss PD, Edmunds WM (1992) Processes controlling acid attenuation in the unsaturated zone of a Triassic sandstone aquifer (UK), in the absence of carbonate. Appl Geochem 7:573–583

Nihlgård BJ (1997) Forest decline and environmental stress. In: Brune D, Chapman DV, Gwynne MD, Pacyna JM (eds) The global environment. Science, technology and management. VCH, Weinheim, Germany, pp 422–440

Painter RB, Blyth K, Mosedale JC, Kelly M (1974) The effect of afforestation on erosion processes and sediment yield. In: Proc Symp Effects of Man on the Interface of the Physical Environment, Paris, IAHS Publ 113, pp 62–67

Penman HL (1948) Natural evaporation from open water, bare soil and grass. Proc Roy Soc London Ser A 193:120–146

Penman HL (1963) Vegetation and hydrology. Tech Comm 53, Commonwealth Bureau of Soils, Harpenden, UK

Rapp C (1992) Effects of liming and N-fertilization on soil chemistry, biomass and nutrient content of fine roots in a mature beech stand in the Solling area (Germany). In: Teller A, Mathy P, Jeffers JNR (eds) Responses of forest ecosystems to environmental changes. Elsevier, New York, pp 829–830

Reuss JO, Cosby BJ, Wright RF (1987) Chemical processes governing soil and water acidification. Nature 329:27–32

Richards LA, Gardner WR, Ogata G (1956) Physical processes determining water loss from soil. Soil Sci Soc Am Proc 20: 310–314

Roberts JM (1983) Forest transpiration: a conservative hydrological process. J Hydrol 66:133–141

Robinson M, Gannon B, Schüch M (1991) A comparison of the hydrology of moorland under natural conditions, agricultural use and forestry. Hydrol Sci J 36:565–577

Rosenqvist IT (1980) Influences of forest vegetation and agriculture on the acidity of fresh water. In: Pfafflin JR, Ziegler EN (eds) Advances in Environmental Science and Engineering. 3:56–79

Ryan MG, O'Toole P, Farrell EP (1998) The influence of drought and natural rewetting on nitrogen dynamics in a coniferous ecosystem in Ireland. Environ Pollut 102:445–451

Seip HM (1980) Acidification of freshwater – sources and mechanisms. In: Drabløs D, Tollan A (eds) Ecological impact of acid precipitation. In: Proc Int Conf, Sandefjord, Norway, 11–14 March, Oslo, pp 358–366

Stanners D, Bourdeau P (1995) (eds) Europe's environment: the Dobris assessment. European Environment Agency, Copenhagen

Stevens PA, Hornung M (1988) Nitrate leaching from a felled Sitka spruce plantation in Beddgelert Forest, North Wales. Soil Use Manage 4:3–9

Stewart JB (1968) Evaporation from forests. Rep 3, Institute of Hydrology, Wallingford, UK

Thompson N, Barrie IA, Ayles M (1981) The Meteorological Office rainfall and rainfall and evaporation calculation system: MORECS. Hydrol Mem no 45, Meteorological Office, Bracknell, UK

Tietema A, Riemer L, Verstraten JM (1992) Nitrification of acid forest soils in the Netherlands; vertical distribution, regulating factors and implications for soil acidification. In: Teller A, Mathy P, Jeffers JNR (eds) Responses of forest ecosystems to environmental changes. Elsevier, New York, pp 763–764

United Kingdom Acid Waters Review Group (1988) Acidity in United Kingdom fresh waters. United Kingdom Acid Waters Review Group, 2nd Rep, Her Majesty's Stationery Office, London, 69 pp

van der Voet E, Kleijn RK, Udo de Haes HK (1996) Nitrogen pollution in the European Union – origins and proposed solutions. Environ Conserv 23:120–132

Verhoef HA, Dorel FG, Zoomer HR, Meinster S (1992) Effects of nitrogen deposition on nutrient cycling and soil biota in coniferous forest soils in the Netherlands: a microcosm approach. In: Teller A, Mathy P, Jeffers JNR (eds) Responses of forest ecosystems to environmental changes. Elsevier, New York, pp 715–716

Virzo De Santo A (1992) Denitrification in forest soils. In: Teller A, Mathy P, Jeffers JNR (eds) Responses of forest ecosystems to environmental changes. Elsevier, New York, pp 208–222

Zhang M, Geng S, Smallwood KS (1998) Assessing groundwater nitrate contamination for resource and landscape management. Ambio 27:170–174

DOI 10.1007/s100400100148