Forestry and the Ecology of Streams and Rivers: Lessons from Abroad?

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Summary

Forests are a natural part of our environment. However, with afforestation representing a major land-use of the upland regions, the possible interactions with waters flowing or draining from these areas merits examination. Several studies from various northern temperate regions have shown that afforestation on poorly buffered soils and in areas of high atmospheric pollution and marine salt influence, can result in profound changes to surface water quality and to the ecology of aquatic systems. The preparation of land for planting and subsequent development of a large canopy of trees can also result in significant changes in water budgets, stream hydrographs and water yields, in comparison to unafforested moorland. Attention has always been focused on the geologically sensitive areas that are negatively affected, but on well buffered soils less significant changes in stream chemistry, due to afforestation, may be expected. However, information from such areas is scarce. In Ireland little research has been carried out on the interactions between afforestation and aquatic systems, and forestry guidelines for protection of fisheries have been drawn up based on information mainly from the United Kingdom and elsewhere. This extrapolation from abroad may not be universally appropriate, as in many parts of Ireland, the soils and geology are generally well buffered, atmospheric pollution (by European standards) is low and the ecology of the systems is somewhat different from that in other countries. This review outlines the nature of changes to freshwater systems that have been found in geologically sensitive areas, but also stresses that extrapolation of results from such afforested areas to all areas under afforestation in general must be undertaken with care. Considerable further work is needed in Ireland and it is hoped this will lead to a better understanding of the interaction between forestry and water resources.

Introduction

Hynes (1975) in his famous essay 'The stream and its valley', recognised the critical link between streams and the surrounding landscape (the catchment), which through exchanges of water, matter and energy can strongly influence the biota of streams and rivers. Features of the catchment such as geology, soils and type of vegetation and land use can thus determine the physicochemistry and ecology of the streams and rivers which drain them. In this context, two major types of land use have been recognised as influencing surface water quality in recent years: agriculture and forestry. Because most afforestation has occurred in upland areas (Parry and Sinclair, 1985), and many major river systems either rise in forests, or receive drainage from planted areas, the interactions between forestry and stream ecosystems have received the most attention and are the subject of the present review.

Forests form a natural part of our environment. Certain tree species are native to Ireland, in particular oak, whilst other exotic species e.g. Sitka spruce and lodgepole pine have been introduced for commercial afforestation and timber production. For comparison on a geographical scale, afforestation covers an area of 7.3% in England, 11.6% in Wales and 12.6% in Scotland, with about 23% of U.K. uplands being planted with conifer forests (Avery and Leslie, 1990). In Nordic countries forestry represents 76% of the land area in Finland and 68% in Sweden. In Ireland however, only 7% of the land is afforested, though planting has increased in recent years and is projected to increase further in the future (Anon, 1991).

Much of the European research on the interactions between forestry and stream ecosystems have been carried out in the United Kingdom (U.K.A.W.R.G., 1988), in particular in Scotland and Wales, although many other studies have been carried out in northern temperate areas such as the USA (e.g. Likens, *et al.*, 1977), Scandinavia and Canada. In Ireland there has been little research on the interactions between forestry and aquatic systems, (see Allott *et al*, 1990) though a major joint research programme recently involved University College Cork, in Munster, Trinity College Dublin in the West, and University College Dublin in Wicklow, on the interactions between forestry and running water systems, (see Giller *et al.* 1993). Thus for the purpose of a review at this time, it is instructive to examine experiences elsewhere and to consider how these results may apply to Ireland.

One of the areas of research that has received most attention is the potential exacerbating effect of afforestation on stream water acidification. Water is a major natural resource and the possible consequences of afforestation on surface water quantity and quality, and on the aquatic ecology of rivers and streams, particularly fisheries, has been a cause for concern. In some areas, predictive models have been produced in an attempt to simulate the potential effects of afforestation under certain conditions; whether these models as they stand will be of use in helping us predict what interactions may occur in Ireland remains to be seen.

Ormerod *et al.* (1991a) identified a number of possible pathways of effect in which the influence of forestry operations on running waters could be summarised (Figure 1). These could be classified as physical, chemical and energetic effects. However in reviewing the interactions between forestry and running water systems it is necessary to state, in the first instance, that the major documented impacts of afforestation on water quality have only occurred under certain conditions. For example, planting of trees in certain geologically sensitive types of geology which have low buffering capacity (Table 1) and in areas receiving high atmospheric pollution and/or under

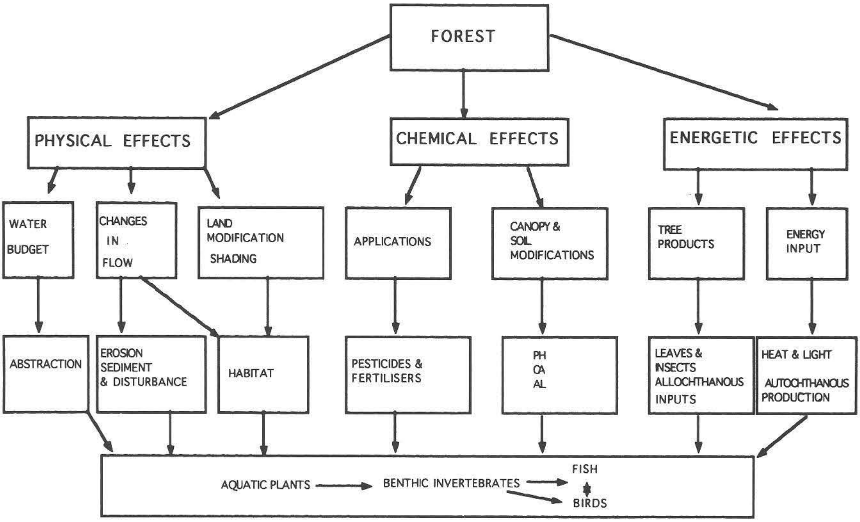

Figure 1: Selected pathways of influence by forest on aquatic fauna (modified from Ormerod *et al.*, 1991).

the influence of marine salt deposition, may have an effect on surface water quality.

In this present review we have concentrated attention on some of the most important effects. Not all the pathways have been fully elucidated yet

Table 1: Classification scheme for the sensitivity of areas of Newfoundland and Labrador based on regional geology. (Anon, 1986)

Class	Relative Sensitivity	Geology
1	Low	Extensive areas of limestone and dolomite
2	Low to moderate	Sedimentary rocks, containing widespread calcium and magnesium carbonates
3	Moderate	Volcanic terrains; major mafic igneous complexes
4	Moderate to High	Quartz-feldspar gnesses; sedimentary rocks poor in calcium and magnesium carbonates
5	High	Granites and related rocks

and this summary represents the major components of the interaction as described from other European and North American research and considers the appropriateness of applying these findings to the Irish situation.

Effects on Chemical Parameters and Surface Water Quality.

In certain geological areas of Wales and Scotland (i.e. those with poor buffering capacity (Table 1), and under the influence of atmospheric pollution), streams and rivers draining conifer afforested catchments have been found to be more acidic and/or contain higher aluminium concentrations than those in adjacent moorlands with similar soils and geology (Harriman and Morrison, 1982; Stoner *et al.* , 1984; Harriman and Wells, 1985; Stoner and Gee, 1985,). The factors and pathways of change in surface water quality due to afforestation have been reviewed by Gee and Stoner (1989) and are summarised in Figure 2.

Trees, both native and exotic, in upland regions scavenge atmospheric anthropogenic pollutants (sulphur dioxide and nitrous oxide), and, near coasts, sea salts from the air. Conifers however, are known to have a greater scavenging capacity than other types of trees (Gee and Stoner, 1989, see Table 2). Pollutants concentrate on the leaves and needles and change the concentration and composition of ions in rainfall and other forms of precipitation (bulk deposition) that reach the trees.

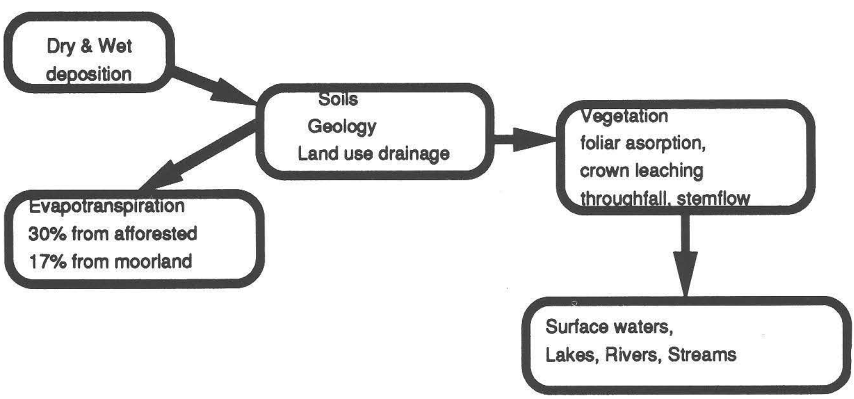

Figure 2: A simplified schematic representation of the factors affecting surface water quality (redrawn after Gee and Stoner, 1989)

Table 2: Comparison of mean values of three chemical parameters in throughfall of bulk deposition under spruce of two ages and oak. Annual range in parentheses (after Gee and Stoner, 1989).

	pH	SO_4 uequiv l^{-1}	Cl uequivl $^{-1}$
Bulk Deposition	4.6 (3.4-7.1)	54 (21-240)	145 (128-873)
Oak	4.7 (4.1-6.1)	115 (39-444)	242 (56-451)
12 year old spruce	4.27 (3.7-6.0)	144 (37-1181)	186 (85-958)
25 year old spruce	4.32 (3.6-5.9)	296 (51-1512)	268 (56-592)

This precipitation, when it reaches the soil by throughfall through the forest canopy, is often more acidic and where the buffering capacity of the area is poor and especially when the rainwater is acidic, this can in turn lead to an acidic runoff and acidification of stream water within an afforested catchment. This process can be illustrated clearly by studies in Llynn Brianne in Wales. The degree of effect, if any, of the introduced ions depends on the water chemistry of the surface waters which in turn relates to the underlying geology of the catchment. In areas of 'soft waters' (<15 mg/l Ca CO_3) the affects can be profound, whereas in areas where the water is 'hard' (>15 mg/l Ca CO_3) the effects are minimal (Jones, 1986). Acidification, (as dry deposition), may be constant over time in heavily polluted environments, but the changes in stream chemistry on poorly buffered soils generally follow a seasonal pattern and may be episodic due to changes in precipitation. For example Gee and Stoner (1989) describe a pH change from 6.2 in summer to 3.9 in winter in streams on poorly buffered soils draining forestry. Similarly calcium and aluminium (particularly labile monomeric) concentrations changed, with calcium levels dropping by three orders of magnitude and aluminium concentrations increasing by almost ten orders of magnitude. In the same way as seasonal episodes can cause changes in water chemistry, short episodes over a 24 hour period may also result in significant changes in surface water chemistry. In Welsh studies, pH changes from 6.0 to 4.0 and an increase of aluminium concentration of 1 mg/l, have been recorded within an eleven hour period after a snow melt (Gee and Stoner, 1989). Studies in the west of Ireland have shown similar dramatic drops in pH following heavy rain (Allott, *et al.*, 1990; Giller *et al.*, 1993).

Sea salt may also exacerbate the acidification process (Allott, *et al.*, 1990). This process of acidification occurs as a result of ion exchange in the soil, leading to an export of protons from the soil. The acidification process is due to sodium ions exchanging for hydrogen ions which leave the system accompanied by the chloride which is highly mobile and present in abundance in the soil. In this process too, aluminium ions may be leached, as they would under the influence of anthropogenically derived sulphate acidification.

A further source of chemical changes to aquatic systems is through the application of various chemicals used during afforestation and agricultural practices which can be biomagnified up the trophic system in streams. Ormerod and Tyler (1992) and O'Halloran *et al.*, (1993) have shown high levels of pesticides in the eggs of Dippers from upland areas, a species of bird that is exclusive to the riparian habitat and feeds on the stream invertebrates (see below). Herbicides which are often used during forestry management, may be washed into the stream systems and hence affect the water quality, its ecology and resource value. Clearly the use of herbicides and similar pest control measures should be notified to the statutory water bodies and the effects on non-target organisms should be monitored closely (Ormerod, *et al.*, 1987a).

Finally, dramatic chemical changes to stream waters can occur following deforestation/clear felling irrespective of whether the system is under the influence of atmospheric pollution or not and whether it is on sensitive or non-sensitive geologies. For example Likens *et al.* (1977) reported a significant increase in the concentration of a range of nutrients in streams in the Hubbard Brook U.S.A. following clear felling of hardwoods.

In summary, surface water quality from afforested catchments on poorly buffered soils in polluted environments can suffer from significant changes in ionic composition with an increase in acidity and an increase in some metal ions. Other forestry practices can also alter stream chemistry. The consequences for the ecology of aquatic systems can be profound when significant changes occur (see below). However, before discussing the ecology in detail it is necessary to review the physical effects of catchment afforestation on the streams and rivers.

The Influence of Afforestation on Physical Parameters

Different stages of forestry practice, such as ground preparation, road construction, drainage, clear felling and general forest management can influence the physical nature of the stream and hence its ecology. This can be via temperature and light attenuation changes, sediment yields, and changing stream hydrographs (Salo and Cundy, 1987). In the following section we consider some of these factors.

Temperature and light attenuation: As water flows downstream, its temperature will continue to change as a result of the influence of several factors that make up the heat balance of water. The net gain or loss of heat by a stream as it moves through a forest is the sum of the net radiation, evaporation, convection, advection and conduction (Brown, 1983). In afforested catchments, solar radiation is intercepted by the canopy of the stream side vegetation. The net solar radiation beneath a continuous canopy may be as low as 15% or less than that of an unshaded stream (Brown, 1983). This change in solar radiation will have serious consequences for the primary production and hence the biology of the streams (see below). However, it is important to point out that because of water's relatively high specific heat, seasonal and daily temperature changes of forested streams are relatively small and gradual (Beschta *et.al.*, 1987), thus in colder upland areas the canopy may buffer the effects of fluctuations.

Following clear felling, changes in the stream temperature and degree of solar radiation can be dramatic. Concern over the altered temperatures after logging stream side vegetation usually focuses on the inevitable increase in maximum temperatures observed during the summer (Beschta *et al.*, 1987). This focus in the U.S., in particular, is very much a result of a toxicological perspective on temperature changes that is prevalent in fisheries research. In the majority of cases however, stream temperatures in deforested watersheds, while invariably warmer than they were in the forested state, do not approach the tolerance limit of the resident fish species (Beschta *et al.*, 1987). The increased levels of solar radiation reaching the stream in a clear felled area will undoubtedly lead to an increase in the stream primary production and this contributes to some of the changes to the ecology that have been reported (see later).

Sediment: Nearly all forestry operations create some degree of soil disturbance (Everest *et al.*, 1987). During the preparation of land for afforestation, road construction and eventual harvesting, sediment from soils is released and washed into streams. Much research has been carried out in the U.S. on the yields of sediment from catchments under these different activities (for review see Ice 1979, Brown, 1983; Sidle *et al.*, 1985, Everest *et al.*, 1987). The sediment in these streams can be of two types, suspended sediment and bedload. These two types of sediment may effect the ecology through damage to fish gills, submersion following a sediment release or by changing habitat structure e.g. spawning and foraging grounds for fish (Everest *et al.*, 1987). At the early stage of planting, ground preparation and drainage will influence the rate of sediment release. Although most mass soil movements appear to be initiated by large rainfall or runoff events, road construction and the roads themselves can provide a major source of sediment in afforested catchments (Everest *et al.*, 1987). The relative importance of roads and clearcuts in accelerating sediment production can vary significantly from one area to another and through time in a single area. The large impact from a small road can equal or exceed the smaller impacts of clear cutting over a larger areas (Swanson and Dyrness 1975). Ice (1985) in his studies of 13 areas in the U.S. found that in 11, road construction contributed to more than half the sediment movement. From the above discussion it is clear that sediment production arises from a complex series of processes which will be influenced by stochastic events such as flooding and wildfire as well as by man's influence through catchment manipulation.

Water yields: Any type of afforestation (natural or exotic) can have a major influence on the catchment's hydrological characteristics and thus influence water yields significantly in comparison to unafforested catchments. In the early stages of afforestation there can be significant changes in stream flow, with increased run-off, sharper storm hydrographs (stream discharge levels over time) and the significant increase in sediment yields described earlier. Once the catchment is covered by closed canopy forestry, the effects of enhanced drainage become subordinate to the processes of soil drying and water loss to the catchment by evapotranspiration (Hornung and Newson, 1986, see below). Comparisons between adjacent catchments of upper River Severn in Wales (afforested) and upper River Wye catchment on Plynlimon, Wales (unafforested), shows a reduction in runoff of between 20-30% in the Severn catchment. Normal flood events tend to reach lower peak discharge levels and droughts may be prolonged due to the increased water storage capacity beneath the forest canopy and in the forest soil (Hornung and Newson, 1986). Two main processes are responsible for these changes in water yields: transpiration loss and evaporative loss of intercepted rain due to afforestation (Gash *et al.*, 1978).

The surfaces of leaves of most plants have numerous small openings called stomata, through which carbon dioxide is taken in for photosynthesis. These stomata also provide the means whereby water is lost from the plant to the atmosphere (transpiration), which results in the drawing of further water and nutrients from the soil. However, a proportion of the rain falling on a forest canopy, between 10-40 per cent, is intercepted by the canopy and evaporates before reaching the soil (Gash *et al.*, 1978), resulting in further losses (interception loss) of water to the catchment.

In summary, the loss of water due to afforestation in a catchment can be very significant for overall water yields in streams and rivers, Edwards and Brooker (1982) estimated that the cost of this water loss in the River Wye catchment in Wales was £0.3 million in 1980. Few other studies have been able to make any predictions along these lines.

In addition to the water yield loss, changes in stream hydrographs and flow changes due to afforestation can have significant influence on the ecology of systems (see Giller, 1990). The changes in water budgets may also lead to different sedimentation rates, flow regimes and erosion, resulting in changes to the stream morphology, ecology and physical properties of river systems. The management implications include potential increased costs due to loss in water yield for abstraction, electricity generation and repairing physical damage to river courses and riparian margins due to flooding.

The Influence of Afforestation on Aquatic Ecology.

In the discussion up to now, we have considered the influence of afforestation on the properties of water. However, water not only provides a resource for extraction, industry and recreation, but is also an important habitat in its own right, in particular in relation to the valuable ecology it supports. Fish, especially salmonids, form part of a complex food web in rivers and streams, and are very sensitive to changes in both the physical (sediment and stream hydrographs) and chemical characteristics of water. However, they are not only dependent on the river water, but also on the large biomass of invertebrate animals which forms the major components of their diet. Rivers are also important for angling and for providing water for fin-fish facilities in the aquaculture industry.

From the earlier sections we have seen that surface water quality can be significantly changed in streams running or draining from poorly buffered afforested catchments compared to unafforested moorland catchments. The ecology of streams and rivers is largely dependent on the energy flow into the system and the quality of the water. This section considers how afforestation, through changes in both surface water quality, and the nature, quality and quantity of detritus (allochthanous matter) inputs into running water systems from the surrounding catchments, can affect the aquatic ecology of streams, especially those running on poorly buffered geologies.

Before examining in detail the influence of afforestation, particularly through exacerbation of acidity in certain areas, it is important to emphasise that naturally acidic ecosystems exist and support their own unique communities that differ from other running water systems. Mean densities and numbers of species of insect larvae are generally higher in naturally alkaline streams compared with naturally acidic streams (northern hemisphere: Hildrew and Townsend, 1987; southern hemisphere: Collier and Winterbourn , 1990). In the same way, Pusey and Edward (1990) have reported low diversity of fish species assemblages in acidic peat ponds in south western Australia and salmonids are frequently absent from naturally acidic systems in the U.K., (U.K.A.W.R.G., 1988). In the systems we are reviewing here, the increased acidity is of anthropogenic origin and exacerbated by afforestation. In such cases the effects are evident at all levels of the ecosystems as outlined below and occur because the rate and degree of change is too rapid to allow community adaptation to the new conditions.

Primary Production: The main reduction in primary production in afforested streams where trees grow close to the stream bank is via a reduction

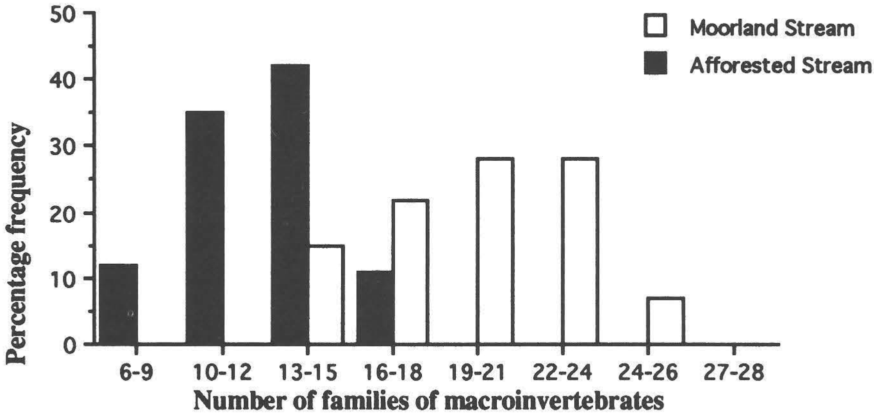

Figure 3: The percentage frequency distribution of macoinvertebrate animals at 25 sites in Wales (after Ormerod *et al.*, 1987b)

in solar radiation due to shading. In addition changes in stream chemistry can affect the plant species in the river system. Upland rivers are usually characterised as being of low productivity (oligotrophic) and afforested streams suffering from acidification generally have a lower diversity of autotrophs than unafforested streams (U.K.A.W.R.G., 1988; Welsh Water 1987). Streams with a closed canopy of forest are dominated by blue-green algae, whilst those draining acidic moorland or where the bank sides have been cleared are dominated by filamentous algae (Gee and Stoner, 1989). Little is known about the bacteria and micro-fauna in streams, but given the importance of these and algae for grazing macroinvertebrates it is likely that these micro-communities are important.

Macroinvertebrates: Field data from a wealth of studies on stream invertebrates, show a close relationship between the hydrochemistry of streams and their invertebrate faunas (e.g. Townsend *et al.*, 1983, Wright *et al.*, 1984, Hildrew and Townsend, 1987, Rutt *et al.*, 1989, Hildrew and Giller, 1994). As mentioned above, naturally acidic waters in general support a lower diversity of organisms than circumneutral and alkaline systems. In the same way, acidic streams draining afforested, poorly buffered catchments are no exception and whilst their invertebrate density may be higher than adjacent moorland (Ormerod *et al.*, 1987a) their diversity is usually reduced e.g. Stoner *et al.*, (1984) (Figure 3).

Some groups of invertebrates such as mayflies, molluscs and crustaceans and some caddis fly larvae are particularly restricted in waters at pH < 5.7 and > 0.1 mg Al/l and these taxa are often scarce in forest streams that lie on sensitive geologies (Raddum, *et al.* 1988).

There is still considerable debate as to whether sensitive taxa such as mayflies are influenced physiologically by pH, aluminium or a combination of both, or whether they are limited by reduced food quantity, quality and temporal availability in acidified streams (Ormerod, *et al.*, 1987a; Willoughby and Mappin, 1988). This food limitation may be either a reduction in leaf litter (allochtanous material) which forms the basic diet of many detritivorous stream invertebrates or by a reduction in primary production, in particular producer species, in the streams (autochtanous production). For example, limits in the production of food rich algae and bacteria (discussed above) may limit the distribution of grazing invertebrates (Ziemann, 1975; Winterbourn *et al.*, 1985). Nevertheless Ormerod *et al.* (1987a) have shown increased mortality of invertebrates independent of food levels in acidic streams in Wales. Physiological effects of low pH have been demonstrated on chironimids (Havas and Hutchinson,1983), mayflies (Willoughby and Mappin, 1988) and crustaceans (Sutcliffe and Carrick, 1973). In experimental studies carried out on five species of macroinvertebrates in the U.S. by Burton and Allan (1986), decreasing pH from 7.0 to 4.0 (no aluminium present) caused significant mortality for all species studied. However, the addition of 0.5 mg/l of aluminium caused additional mortality for only some species indicating that some species are more sensitive to a combination of aluminium and pH than pH alone. In poorly buffered systems the influence of afforestation on invertebrates may seem academic, but these animals provide the corner-stone to the functioning of aquatic ecosystems in both the afforested sections and downstream. In non-sensitive, well buffered geologies, these chemically induced changes in macroinvertebrates do not seem to be apparent, and acid sensitive species are usually found. However, physical changes occurring during various forestry practices can lead to changes in the stream communities. Clear felling for example, leads to alterations in the nature and quality of food available to macroinvertebrates in previously shaded streams (reduced allochthanous and increased autochtanous resources) with consequent changes in the invertebrate communities (Webtser *et al.*, 1983). Complete recovery cannot occur until the quantity and quality of all allochtanous inputs return to the pre-disturbance levels (Wallace *et al.*, 1988), which may take decades.

Fish: Fish are sensitive to changes in food availability (Frost and Browne, 1967; Twomey, 1988) and subtle changes in water chemistry, with many species having different tolerances to acidity and aluminium (Table 3). Stoner and Gee (1985) showed that in mid-Wales, trout were generally absent from conifer forested streams and lakes with hardness < 10 mg/l. More recent data collected by the Welsh Water Authority (1987) have shown that fish are absent from acidic afforested streams, but densities ranged from 0-0.9 m^{-2} in acidic moorlands with low invertebrate density. The question as to whether the acidity *per se* or the aluminium toxicity was limiting the distribution of fish in such afforested streams has received little attention. One study by Ormerod *et al.*, (1987b), using experimental manipulation of streams has shown that the mortality of salmon and trout was dramatically enhanced in the presence of aluminium (0.35 mg/l) at pH 5.0, but mortality was very low at pH 4.3 with no additional aluminium.

Table 3: pH levels at which populations of fish species decline, cease to reproduce, or disappear (various sources).

Species	pH Level(s)
Salmon and Trout	
Rainbow trout	5.5-6.0
Lake trout	5.2-5.5
Atlantic salmon	5.0-5.5
Arctic char	5.0
Brown trout	5.0
Brook trout	4.5-5.0
Others	
Lake whitefish	4.4
Northern pike	4.2-5.2

In Scotland, trout have also been found to be absent from streams draining long established forests and these streams are more acidic, and contain higher concentrations of aluminium than those of neighbouring non-afforested catchments (Harriman and Morrison, 1982). Declines in salmon fisheries have been related to the proportion of afforested sensitive catchments in Scotland by Egglishaw, *et al.*, (1986) and in Wales by Ormerod *et al.*, (1987b).

The effects of acidification and increased aluminium concentrations on the physiology of salmonids have been described by many workers, and are summarised by the U.K.A.W.R.G. (1988). Individual responses to low pH varies according to the dose level, duration of exposure and presence of other ions. Biological variables include species, development stage, size and genetic background. In addition to mortality, many sub-lethal effects have been described including: impaired ion regulation and acid base status, changes in reproductive physiology, developmental effects, de-mineralistion, metal accumulation, reduced growth and changes in behaviour (U.K.A.W.R.G., 1988). There have been a number of other physiological effects reported including a possible impairment of smoltification (Reite and Staurnes, 1987). Potts *et al.*, (1990) suggest that the deaths of adult salmon reported from the rivers Duddan and Esk in U.K. in 1983 were mainly due to inhibition of the sodium uptake system caused by a combination of low pH and high aluminium.

Other vertebrate animals: Other animals associated with rivers include amphibians, birds and mammals. Amphibians occupy a unique niche in temperate regions being dependent on fresh waters at the spawning and larval stages, but species rely on the terrestrial system for the remainder of their life history. Cummins (1986, 1988) has recently shown lethal and sub-lethal effects of low pH and high aluminium on frogs and newts. The European common frog (*Rana temporaria*) has been reported to show poor development at metamorphosis when exposed to pH 4.8 and aluminium concentration 0.8 mg/l (Cummins, 1986). Most research on vertebrates, however has been carried out on aquatic birds, in particular the dipper. Ormerod *et al.*, (1985) and O'Halloran, *et al.*, (1990) have shown that birds on acidic rivers and streams spend more energy searching for food due to the reduced number of invertebrates and that this may not leave sufficient body resources available for breeding. Ormerod *et al.*, (1991b) have shown that breeding of dippers is greatly impaired on acidic streams draining afforested catchments, the poor breeding and absence of the birds was related to both the scarcity of food and to the poor quality of food. When comparing pairs of birds along circum-neutral rivers (> pH 6.0) against those of low pH, dippers on acidic systems breed later, lay fewer eggs and rear smaller broods with slower growing young. Most other research on the influence of afforestation relates to the loss of moorland habitat for breeding wading birds and grouse, though clearly the provision of trees for other species such as hen harriers will prove beneficial (see Avery and Leslie, 1990 for review).

The role of physical effects, such as land preparation, shading, road construction and clear felling will also have an important role in the ecology of vertebrates in fresh water systems. The main effects will be via trophic interactions (such as food provisions) and through habitat provision for foraging and breeding.

Clearly the aquatic ecology of rivers, especially in sensitive geological areas, is very susceptible to changes. In the face of alterations in catchment use, loss of the ecosystem not only results in a major loss of the value of the resource in itself, but in the tourist and associated industry related to angling, aquaculture, conservation and to its intrinsic value.

The Use of Models in Predicting Effects.

Many studies (e.g. Ormerod, *et al.*, 1990), are now using models to predict water quality changes under different atmospheric sulphur emissions, geology and proportion of catchment afforested in attempt to predict the impact of afforestation on poorly buffered soils. One such model is a hydrochemical model called MAGIC (i.e. Model of Acidification of Groundwater In Catchments) which was developed to perform long term simulation of changes in soil and stream water chemistry in responses to changes in acidic deposition. Such models are of use to resource management teams to allow predictions, with a certain degree of confidence, as to the effects of various influences on stream water quality. However, it is important to note that all models are imperfect representations of the real world and often require cautious interpretation. Further research is clearly needed in this area.

Implications for Ireland

As mentioned in the introduction, very little is known about the influence of afforestation on water resources in Ireland, and any negative effects documented so far have been located in geologically sensitive areas and/or those receiving relatively high atmospheric pollution loads (Giller *et. al.,* 1993). In some cases, extrapolations have been made from studies carried out elsewhere. A conservative approach to the potential threat may be considered appropriate (i.e.'better safe than sorry') as in the production of forestry guidelines for the protection of fisheries, but over generalisations should be avoided and caution observed in translating lessons from abroad to the Irish situation, especially given the lack of objective data here. This cautionary approach is based on the environmental differences between Ireland and other countries where the majority of research has been carried out to date. Recent studies partly supported by the forest industry will help to elucidate these differences (see Giller *et al.,* 1993). The main differences are as follows, firstly the level of anthropogenic pollution in Ireland is probably very low, sporadic and with prevailing south-westerly winds the exacerbating effect of afforestation on acid rain is likely to be low. However, because of the maritime nature of the Irish landscape the influence of marine salts and their interaction with afforestation deserves investigation. All the pH problems associated with afforestation mentioned above have only been described for sensitive catchments with poor buffering capacity. However much of the underlying geology in Ireland is well buffered, though acid sensitive regions do exist in the west and in Co. Wicklow.

Other physical effects associated with forestry practice (e.g. establishment and clear felling operations, use of pesticides, shading, road building etc.) have been documented elsewhere and are likely to affect systems in Ireland in a similar way in both sensitive and nonsensitive areas alike. Examination of techniques to moderate adverse physical effects of such practices is required as a matter of some urgency.

Secondly, the species of macro-invertebrates and fish present in streams and rivers vary and are naturally less diverse in Ireland because of the island nature of the country (McCarthy, 1986); thus the ecology of the system is likely to be somewhat different. For these and many other reasons, current research is focusing on gathering information on the above problems. Some of the findings have indicated potential damage (e.g. Allott *et al.*, 1990), whilst other preliminary findings show little affect of afforestation (see elsewhere in this volume). One aspect of investigations which is currently not being fully investigated is the use of predictive models. These models may well help us not only to identify problems, but also direct us to potential solutions.

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Figure 1: Selected pathways of influence by forest on aquatic fauna (modified from Ormerod *et al.*, 1991).