NDP NATIONAL DEVELOPMENT PLAN

Comhshaol, Oidhreacht agus Rialtas Áitiúil
Environment, Heritage and Local Government

western river basin district

# Water Framework Directive
# Western River Basin District

## Programme of Measures

## Forest and water National Study

## Forests and Surface water Acidification Literature Review

November 2008

# LITERATURE REVIEW: FORESTS AND SURFACE WATER ACIDIFICATION

JAMES JOHNSON, EDWARD FARRELL, JAN-ROBERT BAARS, ROBERT CRUIKSHANKS, RONAN MATSON, MARY KELLY-QUINN

# Table of Contents

# 1 Introduction

A Directive (2000/60/EC), establishing a new framework for Community action in the field of water policy, was agreed by the European Parliament and Council and came into force on 22nd December 2000. The Directive, generally known as the Water Framework Directive (WFD) was subsequently transposed into Irish legislation in 2003 (European Communities (Water Policy) Regulations (S.I. no. 722 of 2003)). It requires EU member states to i) implement the necessary measures to prevent deterioration in the status of all bodies of surface and groundwater and ii) where necessary, restore all waters to good ecological and chemical status by 2015.

In 2005, the EPA submitted the National Characterisation Report for Ireland to the EU Commission (Anon, 2005). This included the first risk assessment of anthropogenic pressures based on readily available information. The risk assessment method (Guidance Document GW4) (Anon, 2004) employed in Ireland followed the Source–Pathway–Receptor model. The level of risk was determined by combining data relating to the magnitude and types of pressures present (pressure), the characteristics of the pathway (pathway susceptibility) and of the receptor (sensitivity) in a risk matrix.

It was recognized that large areas of uncertainty existed due to gaps in information and the short timeframe for preparation of the Characterisation Report. To reduce the level of uncertainty and improve knowledge, further characterisation of significant potential pressures, identified by the characterization report as placing water bodies at risk is being undertaken. The objective of this work is to fill in missing information and develop cost effective programmes of measures for inclusion in the River Basin Management Plans.

Each River Basin District has been allocated particular pressures for further investigation. The Western River Basin District has been given, inter alia, the task of assessing the potential impact of plantation forests and forest related activities on surface waters and developing a programme of measures to address any significant risks.

Forestry has been identified as a potential source of diffuse pollution within Ireland's initial Characterisation and Analysis Report (Anon, 2005). Among the potential pressures arising from forestry are increased acidification from plantations in acid-sensitive catchments , erosion on steep catchments due to sedimentation losses resulting from the establishment of new plantations and road construction and eutrophication arising from forest operations on sensitive soil types This document shall review the existing literature relating to the interaction between forestry and surface water acidification with particular reference to the Irish context. It follows the pressure-pathway-and receptor format of the WFD risk assessment model and addresses the following:

a) The processes by which forestry exerts a pressure on surface waters and factors considered to affect the magnitude of the pressure
b) The characteristics which determine catchment and receptor sensitivity and the various approaches used to measure sensitivity
c) Evidence for impacts on the receptor (chemistry and biota)
d) Forest guidelines that aim to protect water quality
e) Existing and potential measures to mitigate acidification
f) Current knowledge gaps

# 2 Identifying the Pressure

Forests may exert an acidifying pressure on surface waters (a) indirectly, through the enhanced transfer of atmospheric pollutants and in some instances sea salts to the catchment or (b) directly, through natural processes which occur within the ecosystem as well as changes to the site as a result of forestry and associated forest practices, such as forest establishment and harvesting.

A number of processes have been suggested to explain the role of forests in increasing acidity and/or aluminium concentration in acid-sensitive surface waters. These include:

1) interception by the canopy resulting in the enhanced transfer of atmospheric pollutants and in some instances sea salts to the catchment

2) uptake of base cations by trees and subsequent removal by harvesting

3) oxidation and mineralization of organic matter resulting in the production of organic acids, sulphate and nitrogen compounds

4) alterations to site hydrology resulting in the reduced residence time of water and reduction in base-flow contribution

5) The short-term release of nitrate following the large-scale felling of forest sites in acid-sensitive catchments

## 2.1 Interception of Atmospheric Pollutants

The role of forestry in the acidification of surface waters is primarily attributed to the interception of atmospheric pollutants (principally compounds of sulphur and nitrogen) coupled with the inability of the substrate soils and geology to buffer the acidity (Jenkins *et al.*, 1990; Ormerod *et al.*, 1991b).

The quantity of atmospheric pollutants deposited at a given site is strongly influenced by the nature of the vegetation layer. Mature forest canopies intercept aerosols and dust particles more efficiently from the atmosphere than peatland vegetation or immature trees (Hornung *et al.*, 1987). This increased capture, the so-called scavenging effect, arises due to the turbulent air mixing above and within the forest canopy and is a function of the stand structure (Fowler *et al.*, 1989). The effect therefore becomes more important as trees grow and the height of the stand increases. The pollutants deposited in this way include nitric acid vapour ($HNO_3$), hydrochloric acid vapour (HCl) and ammonia $NH_3$ (Fowler *et al.*, 1989).

Apart from the nature of the vegetation layer, the quantity of atmospheric pollutants deposited at a given site is strongly influenced by altitude, amount of cloud and bulk deposition. Sulphate and nitrate aerosols that nucleate and accumulate in cloud droplets are deposited in large quantities onto some mid to high elevation forests (300-500m) in the U.K (Fowler *et al.*, 1989). The concentration of ions in cloud water in these areas can be as much as two to ten times greater than that in rain (Crossley, Wilson and Milne, 1992; Neal *et al.*, 2001). High elevation forests are efficient at intercepting these droplets, both in mist and low-lying cloud (occult deposition) because of the increased duration of cloud cover and high wind speeds (Dollard, Unsworth and Harve, 1983). Thus, this deposition pathway may make a large contribution to high elevation sites (Grace and Unsworth, 1988; Fowler *et al.*, 1989). For example, it was estimated that the conversion of a moorland site to coniferous forest (15m tall) in Kielder Forest in northern England, in an area receiving an annual input of 17.5 kg $ha^{-1}$ of sulphur and 12.5 kg $ha^{-1}$ of nitrogen would result in an increase in these pollutant inputs by 30% and 90%, respectively at the site. The additional inputs arise mainly from the interception of sulphate in cloud-water and the dry deposition of nitrate and ammonia on the forest canopy (Fowler *et al.*, 1989). The importance of elevation is also reflected in the UK Forestry Commission guidelines relating to acidification in which some restrictions are placed on afforestation above 300m in areas where critical loads of acidity are exceeded (Forestry-Commission-UK, 2003).

The effects of occult deposition may be exacerbated by high evapotranspiration rates in forest canopies which can lead to chemical concentrations on leaf surfaces that are substantially larger than those measured in cloud drops themselves (Unsworth, 1984). In addition to the contributions of dry and occult deposition, the canopy itself acts as a source of elemental fluxes. This can occur through either leaching from plant surfaces and via the abscission and breakage of plant parts as foliar litter. However, these increases are offset by entry of chemicals through the plant surfaces (Cummins *et al.*, 1995).

Aerosols and particles, whether sea salts or pollutants, once trapped in the forest canopy are subsequently flushed through to the forest floor and surface waters during rainfall events (Gee and Stoner, 1988). As a result, fluxes of ions in throughfall under the forest canopy may be, although not always much greater than in bulk precipitation in the open (Farrell *et al.*, 1993). For example, at a site in Roundwood, Co. Wicklow, the mean annual flux of sulphate in precipitation for the period 1991-1992, was approximately 8 kg S $ha^{-1}$ $year^{-1}$ but over 20 kg S $ha^{-1}$ $year^{-1}$ was measured in throughfall under a Sitka spruce (*Picea sitchensis* (Bong.) Carr.) canopy. Throughfall was more acidic (pH = 4) than precipitation (pH = 4.48) at the same site (Farrell, Cummins and Boyle, 1997).

In general, deposition is enhanced more by conifer forests than by deciduous broadleaved trees (Ranger, 2001). Studies in Llyn Brianne in Wales, reported that $SO_4^{2-}$ concentrations in throughfall were 296 μeq $l^{-1}$ under 25-year old Sitka spruce (*P. sitchensis*) in comparison to 124 μeq $l^{-1}$ under oak (*Quercus robur* L.) Concentrations under 12-year old spruce were 144 μeq $l^{-1}$, which were intermediate between mature Sitka spruce and oak, suggesting an age-related effect (Gee and Stoner, 1988). Interestingly, Farrell *et al.* (1998) reported that interception of sea salts at a deciduous stand of semi-natural sessile oak (*Quercus petraea* (Mattuschka) Lieblein) at Brackloon in the west of Ireland was found to be almost equal to that at a coniferous stand. The unusual interceptive ability of the deciduous woodland may have arisen due to the high proportion of epiphytic vegetation in the stand (Farrell *et al.*, 1998).

Concentrations of nutrients and pollutants may also be higher at the edges of forests than at the forest interior. A study of throughfall fluxes at a deciduous stand in the northeastern United States found that fluxes of sulphur, inorganic nitrogen and calcium were higher in forest-edge zones than the forest interior, suggesting that forest edges act as traps for airborne pollutants and nutrients (Weathers, Cadenasso and Pickett, 2001). It is thought that mature trees at the forest edge, which have greater leaf surface area exposed compared to the middle of a stand, act to trap and concentrate horizontally driven particles and gases or cloud droplets. Thus, in terms of interception potential the shape of the plantation may need to be considered. Similarly, Lindberg and Owens (1992) reported that sulphate and nitrate concentrations in throughfall present at high elevation forest stands were greater at mature forest-edges than in forest gaps occupied by young saplings (Lindberg and Owens, 1992). Studies such as these demonstrate that throughfall fluxes of nutrients and pollutants beneath forest canopies are highly site and element specific and depend on factors such as elevation, aspect, stand age and structure, tree species and the prevailing chemical climate (Erisman *et al.*, 1997).

The capture of anthropogenically-derived acidifying substances has important implications for soil waters and potentially for associated surface waters. In acid mineral soils (pH < 4.2) the main buffering cation is aluminium. Inputs of acidic anions such as sulphate and nitrate are linked to the availability of labile monomeric aluminium species ($Al^{3+}$, $Al(OH)^{2+}$, $Al(OH)_2^{+}$ ) in the soil solution (Khanna and Ulrich, 1984). As a result, labile inorganic aluminium, which is potentially toxic to biota, is released into the soil solution and $H^+$ and aluminium base neutralizing capacity is transported to drainage waters (van Breemen, Mulder and Driscoll, 1983). This may account for the elevated concentration of labile monomeric aluminium found in soil water and associated surface waters in some forests in acid-sensitive areas (Waters and Jenkins, 1992; Hughes *et al.*, 1994; Grieve and Marsden, 2001). For example, in Llyn Brianne in Wales, the concentration of aluminium in soil-water (B-horizon) under 25-year old conifers was 234 $\mu g\ l^{-1}$, several times higher than in moorland (90 $\mu g\ l^{-1}$). The concentration under 12-year old trees (87 $\mu g\ l^{-1}$) was relatively similar to the moorland vegetation (Gee and Stoner, 1988) highlighting a forest-age effect which is discussed later.

In Ireland, the Forestry Ecosystem Research Group (FERG) based at the University College Dublin monitors the chemistry of precipitation, throughfall and soil water at three sites. These are all sites with mature close canopy forestry, one in Roundwood in the Wicklow mountains (Sitka spruce on peaty podzol), one in Ballyhooley in Cork (Norway spruce (*Picea abies* (L.) on podzol) and a third at Cloosh in County Galway (Sitka spruce on blanket peat). At the Roundwood site, the presence of plantation forestry generates effects which may be contributing to a deterioration in soil water quality. Mean annual concentrations of inorganic monomeric aluminium (as $Al^{3+}$) in soil water for the period 1991-1992 at the stand were found to be much greater (1278 $\mu g\ l^{-1}$) than in open site soil water (289 $\mu g\ l^{-1}$). While at the other sites the effects of forestry were not as evident or absent (Farrell *et al.*, 1997).

### 2.1.1 Pollutant identity

The main pollutant of concern until recently has been sulphur due to its established role in surface water acidification. Adverse impacts on streams draining forests have been attributed to the enhanced input of non-marine sulphate in these catchments (Harriman and Morrison, 1982; Waters and Jenkins, 1992; Kelly-Quinn *et al.*, 1996a; Puhr *et al.*, 2000). However, continuing reductions in sulphur emissions have shifted attention to the role of nitrogen. For example, although nitrogen emissions in the U.K. have been declining slowly since the late 1980s, in some areas they now exceed sulphur emissions (NEGTAP, 2001). In Ireland, the spatial maximum deposition of nitrate and ammonium was over 20 kg N $ha^{-1}$ $year^{-1}$ during the period 1985 -1994, as opposed to a maximum of 9 kg $ha^{-1}$ $year^{-1}$ for sulphur (Aherne and Farrell, 2000). As a result, the potential acidification from nitrogen deposition may exceed that of sulphur in certain parts of the country.

In northern temperate regions of the world, forest growth is generally regarded as being limited by below optimum availability of nitrogen, and drainage water from forests and woodlands generally have low nitrate concentrations (Aber *et al.*, 1989). Consequently, nitrogen deposition would not normally be expected to pass through undisturbed forest ecosystems and result in acidification of water. However, in areas of high nitrogen deposition, the availablity of ammonium and nitrate may be in excess of total combined plant and microbial nutritional demand, a condition known as nitrogen saturation (Aber *et al.*, 1989). Nitrogen saturation can be determined by the accumulation of mineral nitrogen in soils or by increased leaching of nitrate below the rooting zone. The significance of this is that nitrate acts as a strong acid anion in aquatic systems, higher concentrations would promote increased acidity and mobilization of inorganic labile forms of aluminium.

Internationally, nitrate leakage from mature forest stands to soil and surface waters has been identified through experimental manipulations (Bredemeier *et al.*, 1998) and in areas of high nitrogen deposition, such as parts of Wales (Reynolds, Ormerod and Gee, 1994; Stevens *et al.*, 1997); Germany (Kreutzer *et al.*, 1998), Norway (Kaste and Skjelkvale, 2002) and the U.S. (Wigington *et al.*, 1996). Nitrification within forest soils depends on soil type. In upland Wales, forested sites on free draining stagnopodzol soils in which nitrification occurs freely have been found to be nitrogen saturated. A study of 25 sites on stagnopodzol soils in north and central Wales by Hughes *et al.* (1994) found that nitrate concentrations in soil water were considerably greater beneath older forest plantations (30 and 45 years) in comparison to younger plantations and acid moorland. Similarly, Stevens *et al.* (1997) found higher nitrate concentrations in soils and stream waters draining older Sitka spruce stands (37 and 53 years) than in younger forests (14 and 28 years) and a moorland site. Although the highest concentrations were found in the 37 year old opposed to the 53 year old forest stand. In some cases, nitrate leaching to soil and surface waters was accompanied by elevated concentrations of inorganic aluminium (Hughes *et al.*, 1994).

Nitrification is not likely to be important in poorly draining soils such as gleys and peats (Hughes *et al.*, 1994). In Ireland, phosphorus is considered the major limiting nutrient for forest plantations. Nitrogen saturation has not been widely determined, but has been considered to occur at a monitored site at Roundwood, Co. Wicklow (Farrell *et al.*, 2001).

It is also important to note that nitrate can be generated internally from the microbial transformation of ammonium. In areas of intensive animal production, ammonium inputs may be significant (Aherne and Farrell, 2000). Ammonia neutralizes acidity both in precipitation and throughfall, forming ammonium (Farrell *et al.*, 2001). However, added ammonium reaching the soil increases soil acidification either through plant uptake of ammonium or the microbial transformation of ammonium to nitrate (Bredemeier *et al.,* 1998). At the site in Roundwood, Co. Wicklow nitrification was an important source of $H^+$, more so than $H^+$ deposition (Farrell *et al.,* 2001).

Although nitrate loss can potentially acidify surface waters this has not been widely reported. In areas of the northeastern United States with acid soils and where nitrogen deposition is high, nitrate leaching under mature northern-hardwood or mixed hardwood conifer stands has been associated with acid episodes in streams (Wigington *et al.*, 1996). In acid-sensitive catchments in Wicklow, it was found that on an equivalence basis, nitrate was often as important as sulphate in contributing to stream acidity (Kelly-Quinn *et al.*, 1996a).

### 2.1.2 Magnitude of the Pressure

The magnitude of the pressure exerted by the scavenging effect of forests depends primarily on (a) the pollutant load and (b) the percentage of catchment forest cover. The extent of the pressure varies with tree species with some species, such as Sitka spruce (*P. sitchensis*), being more effective scavengers of pollutants than others. The pollutant load at a site is further influenced by climatic conditions such as the frequency and magnitude of rainfall events, the level of annual rainfall, prevailing wind direction and air mass circulation patterns as well as site characteristics such as elevation and aspect, tree species, stand age and structure.

## a) Pollutant Load

The magnitude of the pressure exerted by forests on surface waters depends on the atmospheric chemical climate. Much of the work relating to forests and surface water acidification has taken place in upland areas of Scotland, Wales, and northern England which have historically received high pollutant inputs in the form of large volumes of moderately polluted rainfall. However, as a result of reductions in emissions, deposition loads of sulphur have declined to half their previous values (NEGTAP, 2001). For example, in Galloway in Scotland acid inputs from the atmosphere in the period 1986-88 were in excess of 25 kg S $ha^{-1}$ $yr^{-1}$, whereas contemporary deposition is in the order of 9-12 kg S $ha^{-1}$ $yr^{-1}$ (Langan and Hirst, 2004).

In Ireland, air quality and precipitation data measured at Valentia observatory in the south west of the country suggest a similar trend. Over the period (1980-2004) ambient SO2-S and SO4-S levels decreased by about 60% and 40% respectively. Monitoring of nitrogen dioxide ($NO_2$-N) commenced in 1989 and so the data series is less extensive than that for $SO_2$-S and $SO_4$-S and there was no observable trend from the annual average data (Bashir *et al.*, 2006a). No such obvious trends were apparent from the original precipitation data recorded at Valentia; however by investigating the contribution of sea-salt sulphate to the total sulphate levels Bashir *et al.* (2006) concluded that non-seasalt contribution (i.e. anthropogenic component) has decreased since measurements began (Bashir *et al.*, 2006b).

There is also a distinctive gradient of deposition in precipitation from the east to the west coast in Ireland. The concentration and deposition of the major ions in rainfall were mapped for the period 1985-1994. However, it was not possible to quantify dry deposition of sulphur and nitrogen as the data necessary were not available (Aherne and Farrell, 2000). Bulk deposition of sulphur and oxidised nitrogen from anthropogenic sources was low over the western part of the country but was significant in eastern areas. The highest non-marine sulphate deposition occurred in the Dublin and Wicklow mountains, where the annual average ion deposition was 350 eq $ha^{-1}$ $yr^{-1}$. Nitrate concentration and deposition showed similar distributions to non-marine sulphate. Deposition maps showed that the lowest concentrations occurred in the west (annual ion deposition 100 eq $ha^{-1}$ $yr^{-1}$) and the highest in the east (250 eq $ha^{-1}$ $yr^{-1}$) of the country. In addition, ammonium deposition was appreciable in large areas of the south and east, as a result of the relatively high ammonia emissions in these areas. The spatial average deposition of nitrate and ammonium combined was approximately 342 eq $ha^{-1}$ $year^{-1}$ (144 and 238 eq $ha^{-1}$ $year^{-1}$ respectively), while in some areas the spatial maximum reached over 1200 eq $ha^{-1}$ $year^{-1}$. In comparison, maximum deposition of sulphate has been recorded at 584 eq $ha^{-1}$ $yr^{-1}$ (Aherne and Farrell, 2000). As mentioned previously, this suggests that nitrogen is probably the parameter of most concern in terms of surface water acidification in certain parts of the country.

Emissions agreements such as the Gothenburg Protocol (1999) and the Large Combustion Plant Directive (DIRECTIVE 2001/80/EC) have set emission ceilings for pollutants such as sulphur, ammonia, oxides of nitrogen (NO) and dust (particulate matter) and will likely lead to continued reductions in emissions. Consequently, the magnitude of the acidification pressure exerted by forests will change also.

The concentration of pollutants deposited at a site is influenced by climatic conditions such as the amount of annual rainfall and prevailing wind direction. Areas of Scotland (Galloway and Trossachs), Wales (Plynlimon and Llyn Brianne) have historically received high pollutant inputs in the form of large volumes of moderately polluted rainfall. Annual average rainfall in these areas ranges from approximately 1,800 mm in Wales (Stoner, Gee and Wade, 1984) to over 2,000 mm in Scotland (Langan and Hirst, 2004).

Furthermore, the input of non-marine sulphate and nitrogen compounds depends on the direction of prevailing air masses, with elevated deposition of pollutants associated with air masses coming from an easterly direction. Air masses originating over the Atlantic are generally associated with low pollution deposition (Bowman, 1991). Bowman (1991) reported that elevated deposition of pollutants at West coast sites were associated with air masses from an easterly direction, with significant episodes of acid deposition associated only with air masses that have passed over UK or mainland Europe. In contrast, air masses originating over the Atlantic were generally associated with low pollution deposition. Wind sector analysis of the $SO_2$-S and $SO_4$-S data from Valentia observatory in the southwest of Ireland, for the period 1980-2004, shows that the highest concentrations of these species are correlated with winds coming from an easterly and north-easterly direction (Bashir *et al.*, 2006a). In Wicklow, the higher annual concentrations of non-marine sulphate in surface waters were associated with easterly airflows, air masses that have been transported over major emission sources in the UK and mainland Europe (Aherne and Farrell, 2000). Differences in the chemistry of acid episodes in some streams were also attributed to the direction of prevailing air masses (Kelly-Quinn *et al.*, 1996a). Inputs of sulphate and nitrate and resultant acidity were higher in a forested stream, during a period of prolonged easterly airflow in March 1991, than during a period of predominantly westerly airflow in March 1992 (Kelly-Quinn *et al.*, 1996a). Similarly, in Scotland, it has been reported that the deposition of acid pollutants are associated with air masses crossing Scotland, from a southerly or easterly direction (Harriman and Morrison, 1982).

Climate also influences acidification of surface waters through the magnitude and duration of rainfall events. In Ireland, acid episodes generally occur in winter during periods of high flow following heavy rainfall. Forest streams in acid sensitive areas can on occasion respond rapidly to storm events to the extent that pH may fall considerably within a few hours (Kelly-Quinn *et al.*, 1996a; Allott *et al.*, 1997), as determined in studies in Wicklow and Connemara/South Mayo

## b) Percentage catchment forest cover

Much of the work that has examined the role of forests in surface water acidification has taken the form of detailed comparative catchment studies (Harriman and Morrison, 1982; Lees, 1995; Nisbet, Fowler and Smith, 1995; Helliwell *et al.*, 2001). In general, these studies do not examine the impact along a gradient of percentage catchment forest cover. As such, it is difficult to determine a threshold above which adverse impacts are apparent on stream chemistry or biology in acid-sensitive areas. However, a number of regional studies have attempted to clarify the relationship between the extent of catchment afforestation and surface water chemistry across a range of catchment characteristics. In Wales, the extent of forest cover associated with given chemical conditions (pH < 6, total aluminium >80 µg $l^{-1}$) was determined for three categories of acid sensitivity as represented by stream hardness as opposed to alkalinity. All streams with total hardness <10 mg $CaCO_3$ $l^{-1}$ would have been classified as sensitive (Ormerod, Donald and Brown, 1989). Streams with hardness values between 10-15 mg $CaCO_3$ $l^{-1}$ on average, had pH below 6 at 30% upstream catchment forest cover, while all streams with hardness >15 mg $CaCO_3$ $l^{-1}$ were classified as non-sensitive (Ormerod *et al.*, 1989).

In Ireland, a number of studies have examined the interaction between forestry and water quality. These include a national study of the impacts of acid deposition on acid-sensitive lakes and their associated inflow streams between 1987 and 1989 (Bowman, 1991) as well as an examination of the causes of acidification of surface waters in Connemara and South Mayo in the west of Ireland between December 1989 and May 1990. Subsequently, a national study in Ireland, AQUAFOR, examined the interaction between forestry and surface water chemistry and its influence on fish and invertebrates mainly in headwaters (the first 2.5 km of the watercourse from its furthest upstream source) of acid-sensitive areas in Wicklow (Kelly-Quinn *et al.*, 1996a), parts of Galway and south Mayo in the west (Allott *et al.*, 1997) as well as a less sensitive area in south Munster (Giller *et al.*, 1997).

In Galway and South Mayo, most of the sites examined during the period 1990 - 1992 had alkalinity values close to zero or negative during winter months. These sites were predominantly small headwaters or small streams (2nd order) with peaty catchments draining granite and quartzite bedrock. Forestry was considered to increase the acid status of poorly buffered catchments by increasing inputs of acidifying pollutants. A strong correlation was found between the percentage mature coniferous forest cover and stream $H^+$ concentration ($R^2 = 0.674$). However, the study did not determine a threshold of forestry above which adverse impacts would occur on the stream ecology (including invertebrates and fish) (Allott *et al.*, 1997). In Co. Wicklow, 47 sites, mainly on headwater tributaries were sampled from June 1990 – July 1992. Eight sites were considered to have had a biological impact, 7 of which had forest cover

>25%. However, the study did not have sufficient replication to establish if this was indeed a threshold for impact (Kelly-Quinn *et al.*, 1996a).

A more extensive study of less sensitive forested sites in south Munster, mainly in counties Cork, Waterford and Kerry where forest cover within catchments varied from <20% to over 90%, found no correlation between the percentage catchment forest cover and pH or alkalinity (Giller *et al.*, 1997). Forested sites were considered sufficiently well buffered to acidifying inputs to avoid any impact (Giller *et al.*, 1997).

It is unlikely that a simple relationship exists between the amount of forest cover and degree of acidification in acid-sensitive areas. A more complex model is probably required taking into account the structure and location of the forest. Puhr *et al.* (2000) incorporated the scavenging effect of trees of different height using an index based on the mean height of the forest. This method takes into account the scavenging effects of immature trees but gives them a lesser weighting, for example, by using this method, a catchment with 100% cover of 3m high trees is given the same weight as a catchment with 50% cover of forest of 6 m high trees. The study examined the relationship between high flow water chemistry and conifer forest in 95 streams spread over an area of 2000 $km^2$ in Galloway Scotland. The study found that pH levels were lower and concentrations of aluminium were higher in streams draining granite and Ordovician rock catchments with higher mean forest height (MFH) (Puhr *et al.*, 2000). This agrees with studies, which state that mature forest is more effective at scavenging (Hornung *et al.*, 1990c).

In the U.K., in the most recent edition of the forestry and water quality guidelines (2000), the Forestry Commission stipulates a threshold of up to 10% for new planting with conifers (and up to 30% for broadleaves) in areas where critical loads of acidity are exceeded (Forestry-Commission-UK, 2003). This threshold appears to have been derived from work carried out by the Forestry Commission in Wales (Nisbet, 2001). In Ireland, the Irish Forestry Service of the Department of Agriculture and Food uses alkalinity readings in the assessment of afforestation grant applications in potentially acid-sensitive catchments. Where the minimum alkalinity of runoff water is less than 8mg $l^{-1}$ $CaCO_3$ no afforestation is permitted. In areas where alkalinity exceeds 15 mg

$l^{-1}$ $CaCO_3$ afforestation is permitted and if the alkalinity values are between 8 and 15 mg $l^{-1}$ $CaCO_3$, full, partial or no afforestation may be allowed. Sampling must take place on a minimum of four occasions between 1st February and 31st May at intervals of not greater than 4 weeks (Forest-Service, 2004).

## *2.2 Interception of Sea Salts*

Marine ions, sodium, chloride, magnesium and sulphate dominate the atmospheric input to Irish surface waters, particularly in the west (Giller *et al.*, 1993; Aherne and Farrell, 2000). Forests capture marine ions as wet or dry deposition and thus accentuate the influence of the marine environment (Farrell, 1995; Harriman, Anderson and Miller, 1995). During storm events, high salt inputs result in the displacement of other cations by $Na^+$. The associated chloride ion is largely conservative and most of it is quickly leached. As it passes through the soil to associated drainage water it must be accompanied by a cation. In strongly acid soils, $H^+$ and $Al^{3+}$ are the most abundant cations and as a result, concentations of both increase in drainage water. This does not involve any additional acidification of the soil water system, but rather a transfer of acidity from the exchange complex of the soil solution and possibly to surface waters (Farrell, 1995). Empirical evidence for this process may be found by examining the ratio of Na:Cl in soil and surface water (Heath *et al.*, 1992). In precipitation bearing sea-salts, this ratio is close to 1 (0.86). As Cl ions pass through the catchment and Na ions are retained the Na:Cl ratio decreases, leading to a sodium deficit (Heath *et al.*, 1992). The ‘sea-salt effect’ has been shown to cause episodes of acidity in surface waters in Scotland (Langan, 1987; Ferrier *et al.*, 2001), Scandinavia (Wright *et al.*, 1988; Hindar *et al.*, 1995; Larssen and Holme, 2006), North America (Heath *et al.*, 1992) and Ireland (Farrell, 1995; Allott *et al.*, 1997).

Studies demonstrating this effect suggest that very large inputs of neutral salts are required before an ion exchange effect will be observed (Harriman *et al.*, 1995; Hindar *et al.*, 1995). This may be due to the unfavorable thermodynamics of trivalent Al and proton exchange for $Na^+$ on the soil surfaces (Heath *et al.*, 1992). One such episode was recorded at a site in Galway during a period of very strong south-west to west winds coupled with heavy rainfall (Allott *et al*., 1997) (as part of the AQUAFOR study). The concentration of $Cl^{-1}$ in stream water increased by over 2,000 μeq $l^{-}$ (730 to over 3,000 μeq $l^{-1}$). The pH at the forested site dropped from 4.37 to 3.9 while that measured at an unforested site changed from 5.65 to 4.67. This resulted from the displacement of $H^+$ by marine cations in catchment soils as reflected by a large $Na^+$ deficit (> -450 μeq $l^{-1}$). Similar findings have been reported from Scandinavia. Concentrations of $Cl^-$ increased from 200 to over 1400 μeq $l^{-1}$ in a forested stream on the west coast of Norway. Concentrations of $Na^+$ showed a deficit of -208 μeq $l^{-1}$, which was compensated by a corresponding increase in Al and $H^+$. As a result, the pH in the stream dropped from over 5 to 4.45 (Hindar *et al*., 1995).

The effects of marine episodes may be greater in streams draining areas receiving acid deposition and some authors argue that sea-salt acidification is not a natural process but that depletion of soil base saturation by acid deposition is a prerequisite (Heath *et al*., 1992; Evans, Monteith and Harriman, 2001; Larssen and Holme, 2006). Long-term monitoring data from Norway show that a sea salt event of a given size mobilizes less $Al^{n+}$ when the anthropogenic S-deposition is low. This may be due to the effects of acid deposition in leaching base-cations from forest soils, diminishing the supply of buffering cations, namely $Ca^{2+}$ and $Mg^+$ (Alexander and Cresser, 1995). During storm episodes in six streams in western Norway, $Cl^-$ concentrations in streams increased several fold from pre-storm values to 570 μeq $l^{-1}$ during a winter storm. Values of pH dropped and concentrations of $Al^{n+}$ increased due to cation exchange of $Na^+$ ions for $Al^{n+}$ and $H^+$ in the soil. The response was larger in streams draining catchments receiving high acid deposition and in those afforested with spruce as opposed to native birch (Larssen and Holme, 2006).

### 2.2.1 Magnitude and Frequency of Sea-Salt Episodes

Sea-salt episodes occur on Atlantic coasts and are associated with precipitation during periods of strong westerly or south-westerly airflows during winter. It is unclear how frequently major storm events which result in acid episodes occur. In both Norway and the U.K. the frequency of westerly or south-westerly storms has been shown to vary substantially from year to year with maxima associated with high winter values of the North Atlantic Oscillation Index (NAOI) (Evans and Monteith, 2002; Hindar *et al.*, 2003). The NAOI is a measure of the difference in sea-surface pressure between the Azores and Iceland (Hurrell, 1995). As a result, long-term correlations have been observed in UK acid waters between the winter NAOI and concentrations of both marine ions such as chloride ($Cl^{-1}$), and displaced non-marine cations such as $Ca^{2+}$, $Al^{n+}$ and $H^{+}$. Furthermore, evidence has recently been presented to suggest that seasalts may increase the adsorption of non–seasalt $SO_4^{2-}$ in certain soils (Hughes *et al.*, 1994; Harriman *et al.*, 1995). These relationships suggest that, in addition to episodic effects, varying sea-salt inputs can generate cyclicity in runoff chemistry on a decadal time scale, potentially influencing acidity trends over prolonged periods (Evans and Monteith, 2002). These variations may mask underlying anthropogenic driven trends or create spurious trends. The approximately decadal time scales over which these cycles tend to operate mean that monitoring will need to operate over long periods in order to adequately assess the impact of anthropogenic factors on surface water quality (Evans, 2005).

Surface water acidification associated with sea salt episodes may last for a period of days, however the recovery of the Na:Cl equilibrium in drainage and surface waters may take several years (Farrell, 1995; Hindar *et al.*, 1995). This may be due to the slow reloading of the soil profile with $Al^{n+}$ and $H^{+}$. Reloading the soil profile with $A1^{n+}$ and $H^{+}$ back to prestorm values will affect the catchments ability to mobilize these ions during future sea salt episodes. More frequent episodes will probably result in less acid and thereby less toxic stream-water due to incomplete re-acidification of the soils (Hindar *et al.*, 1995).

Few studies have investigated the potential for sea-salt episodes to impact on aquatic biota. Bowman (1991) reported that short pulses of increased acidity, associated with sea salt episodes did not have an adverse impact on the fauna of acid sensitive streams in Lough Veagh (Donegal) or Maumwee (Galway) in the west of Ireland (Bowman, 1987 and 1991). However, a number of studies in western Norway, have reported that major storm events have led to episodic mortalities of brown trout (*Salmo trutta* L.) in coastal watersheds (Teien *et al.*, 2004a).

## 2.3 *Uptake of base cations and subsequent removal by harvesting*

The removal of base-cations in forest harvesting may lead to acidification of acid soils on the decadal to centennial time scale (Alexander and Cresser, 1995). The magnitude of this particular impact is hard to gauge as the mechanisms which offset the acidification are poorly quantified. These include an increase in base cation weathering beneath the trees and enhanced atmospheric base cation inputs due to scavenging by the forest canopy (Neal and Reynolds, 1998; Reynolds and Stevens, 1998; Reynolds *et al.*, 2000). In addition, it has been pointed out that the uptake of bases by roots, though involving a balancing output of $H^+$, does not involve the mobilization of any strong anion, which would transfer acidity to drainage waters(Nilsson, Miller and Miller, 1982). However, long term monitoring is required to determine whether repeated rotation cycles of forestry could lead to acidification.

## 2.4 *Oxidation of organic matter resulting in the production of organic acids,*

Naturally occurring organic acids make significant contributions to water acidity in peaty catchments (Oliver, Thurman and Malcolm, 1983). Increased drying of soil and altered drainage increases the oxidation of organic matter and generates carboxylate anions. These anions and an equivalent acidity are released from ion-exchange reactions in the soil into the drainage water, where further $H^+$ may be released through dissociation (Oliver *et al.*, 1983).

As a result surface waters draining forested, peaty-catchments may have higher concentrations of organic acids, even in the absence of atmospheric pollution (Kelly-Quinn *et al.*, 1996a; Allott *et al.*, 1997). This was evident in the AQUAFOR studies carried out in Wicklow and Connaught in the early 1990's. In Galway, in the west, organic acids contributed to 9 of 12 acid events recorded in a peaty catchment. In some instances the contribution of organic acid ions in forested streams was twice that of a comparative non-forested catchment (Allott *et al.*, 1997).

However, organic acids have a weak dissociation constant and may contribute to acidity but they are unlikely to account for the higher acidity and concentrations of labile monomeric aluminium in some forested streams. This is because organic acids form complexes with inorganic aluminium in the non-labile form, thereby mitigating its toxicity to biota (Driscoll *et al.*, 1980).

## 2.5 Site Preparation

Up to the mid-1980s, the establishment of a conifer forest plantation in Ireland involved ploughing prior to planting which in many cases resulted in considerable alterations to the existing landscape. The resultant disruption of the surface soil layers results in increased aeration and drying, with the potential for increased mineralization of organic matter and consequent increases in losses of NH, and/or NO,$^{-}$ as well as sulphate to drainage waters (Hornung *et al.*, 1995). However, the potential of this mechanism as a cause of acidification in acid-sensitive areas has not been established.

## 2.6 Alterations to site hydrology

Alterations in site hydrology by plantation forestry that result in more rapid run-off contribute to a decrease contact time for buffering reactions to take place (Waters and Jenkins, 1992). Where improved drainage permits the rapid transit of storm runoff, the difference between storm and base flow chemistry is more marked (Langan and Hirst, 2004). In the AQUAFOR study in Co. Wicklow, forested streams were more acidic than moorland streams, even when atmospheric inputs were low. The loss in catchment hardness and alkalinity recorded in these streams was partly attributed to the improved drainage leading to higher discharges in these streams (Kelly-Quinn *et al.*, 1996a). However, drainage at site is highly dependent on site characteristics such as slope, soil type and geology. Furthermore, practices carried out as part of forestry establishment have changed considerably since the forests within these studies were carried out. For example, in Ireland, since the advent of the forestry and Fisheries Guidelines in 1991, the direct discharge of a forest drain into a receiving water is forbidden.

## 2.7 Felling

The short-term release of nitrate that may follow the large-scale harvesting of some forest sites has been cited as an additional acidification threat within acid-sensitive areas in the U.K. (Forestry-Commission-UK, 2003). An extensive study in upland areas of Wales and northern England examined the effects of conifer harvesting and planting on water quality. The areas examined were acid-sensitive, located in areas of Palaeozoic sedimentary rock generally mudstones, slates and shales. Soils in these areas were acidic or acid-sensitive and include peats, peaty podzols and gleys and brown podzolics. Two types of site were monitored, those with areas varying between 50 and 300 ha and small catchment studies with areas less than 15 ha (Neal *et al.*, 1998). Felling ranged between <25% to 100% of catchments.

The major responses to felling included; firstly, an increase in nitrate leaching for a period of up to four years post felling (Neal *et al.*, 1998; Neal *et al.*, 2003; Wang *et al.*, 2006), secondly, a decrease in sea-salt components such as sodium and chloride derived from atmospheric sources, due to reduced scavenging and evapotranspiration of mist and aerosols from the atmosphere by the removal of the canopy (Neal *et al.*, 1998; Neal *et al.*, 2004b). Thirdly, the perturbation in water quality associated with felling receded after a few years as nitrate uptake by tree re-establishment and vegetation growth increases and the concentrations of the major acid-anions decreased. There was little evidence to suggest that nitrate generation following felling gives rise to stream acidification at the catchment level (Neal and Reynolds, 1998). In one small (13 ha) first order catchment annual average alkalinity and pH values decreased from -20 to -30 $\mu eq\ l^{-1}$ and from 4.7 to 4.5 for two years following clearfelling. Subsequently, the acidification was reversed after five years post felling and the system had higher pH and Gran-alkalinity (4.9 and 15 $\mu eq\ l^{-1}$ ) than before felling occurred (Neal *et al.*, 2003).

In Ireland, there is little documented evidence to assess whether an acidification effect occurs following felling. At Cloosh in Galway, the effects of felling on stream water chemistry were examined at two small (1ha) and one larger catchment (approx, 100 ha) located in an area of peatland. The two small catchments were clearfelled and one third of the larger catchment felled in 1999. The study found no discernible effect of felling on streamwater pH, alkalinity or aluminium concentration (Cummins and Farrell, 2003).

The limited impacts of felling on surface water quality in the U.K. and Ireland is likely the result of the practice of phased felling and the use of other protective measures such as buffer zones and silt traps. Forestry and Water Quality guidelines recommend that coupe size be determined by the sensitivity of the site (Anon, 2000b). By using this system, any change arising from felling is diluted by unfelled parts of the catchment or by increased nitrate uptake by other parts in recovery following felling (Neal *et al.*, 2004a). Rapid revegetation of harvested areas is a major factor in reducing the duration of the nitrate leaching pulse (Neal *et al.*, 1998).

# 3 Pathway Susceptibility

The potential for ecological damage by forests depends on the sensitivity of the catchment (buffering capacity) which is expressed primarily by a measure of either the alkalinity or acid neutalizing capacity of the run off water. Alkalinity is imparted to the water through contact with the soils and bedrock of the (Hornung *et al.*, 1995). Also catchment size and hydrology have a bearing on the susceptibility of running waters to acidification.

## *3.1 Catchment Soils and Geology*

Underlying geology, soil type and the extent of atmospheric pollution appear to be of principal importance in determining whether acidification will affect receiving water bodies, as little impact has been found in areas of low atmospheric pollution and in areas draining geologies with high buffering capacity. The available literature indicates that water bodies susceptible to acidification are located in catchments dominated by slow weathering bedrock such as granite and quartzite with shallow carbonate free soils as well as areas of sandy, siliceous soils and highly weathered old leached soils (Hornung *et al.*, 1990b). The lakes and rivers associated with such areas are relatively simply characterised by their low buffering capacity as indicated by alkalinity or acid neutralising capacity (ANC) values close to zero and low electrical conductivity. Atmospheric deposition of acidifying pollutants such as sulphate, nitrate in these waters results in a reduction in surface water pH values (<5.5).

Much of the work that has examined the role of forestry in surface water acidification has been carried out in upland areas of Wales such as Plynlimon and the River Twyi catchment and areas of south and central Scoland such as Galloway and the Trossachs. For example, in the River Twyi catchment in Wales, acid-sensitive streams drain areas of Ordovician and Silurian mudstones slates and shales, Soils in these areas are oligomorphic peats on hill tops, ferric stagnopodzols on upper slopes and brown podzols on lower slopes (Stoner *et al.*, 1984).

In Scotland, Puhr *et al.* (2000) examined the interaction between coniferous plantations and streamwater chemistry on a number of acid-sensitive sites in areas of granite, Ordovician and Silurian rock. The relationships between pH and aluminium and forestry were strongest for sites draining granitic rock and Ordovician rock and weakest for those in Silurian rock catchments. Analysis of chemical differences between the most heavily and least afforested sites suggests that forestry-related increases in acidity were greatest for sites in granitic catchments and smallest for those draining Silurian rocks (Puhr *et al.*, 2000).

In Ireland, acid-sensitive surface waters have been found in areas of the Wicklow mountains and along the west coast of Ireland (Bowman, 1991). In Wicklow, acid-sensitive streams are found in areas of Paleozoic sedimentary rock overlain by mineral soils such as brown podzolics and gleys as well as areas of granite with blanket peat and peaty soils (Kelly-Quinn *et al.*, 1996a). Acid-sensitive streams in these areas sampled during the period June 1990-July 1992, were characterized by low alkalinity, < 200 μeq $l^{-1}$ (< 10 mg $l^{-1}$as $CaCO_3$) and low non-marine hardness values (<120 μeq $l^{-1}$)(<6 mg $l^{-1}$ as $CaCO_3$). The streams draining Paleozoic sediments with mineral soils were slightly better buffered than streams with high concentrations of organic acidity (DTOC >7 mg $l^{-1}$). Values of pH rarely fell below 5.0 in these streams, however mean concentrations of labile monomeric aluminium were often above 40 – 50 μg $l^{-1}$ in many of the forested streams studied. This in combination with low concentrations of calcium (<1 mg $l^{-1}$ ) within the pH range of 5.0 – 5.5 represents potentially toxic conditions for aquatic biota (Turnpenny *et al.*, 1987).

Extensive acid-sensitive areas occur in the west of Ireland, in parts of Galway and Donegal. These are characterised by base-poor, slow weathering geologies such as granites and quartzite and are overlain by blanket peat and/or peaty podsols. The surface waters in these areas are low in alkalinity (< 10 mg $l^{-1}$ as $CaCO_3$) and consequently have a poor buffering capacity (Allott *et al.*, 1990; Bowman, 1991; Allott *et al.*, 1997). Here the contribution of organic acids is often significant. In addition, a survey of over 200 lakes in acid-sensitive areas carried out in 1997 reported that the organic acids were the dominant source of acidity in over half those surveyed (Aherne, Kelly-Quinn and Farrell, 2002). The significance of this is that organic compounds are considered to complex with aluminium forming the inorganic non-labile form thus rendering it non-toxic to biota (Driscoll *et al.*, 1980).

In Munster, a study of 45 sites in counties Cork, Kerry Waterford and south Tipperary during 1991 and 1992, investigated the potential for a forest effect on surface water chemistry. The sites were located mainly in areas of Old Red Sandstone with some shale and conglomerate (Giller *et al.*, 1997). The study reported that half the sites had mean alkalinity values of less than 10 mg $CaCO_3$ $l^{-1}$ yet there was no evidence of an acid-related effect on biota due to the presence of forestry (Clenaghan *et al.*, 1998; Lehane *et al.*, 2004). The majority of sites had pH values greater than 6 on most occassions. Only three sites could be considered acidic (i.e. the pH value approached 5): two headwaters of the Argalin river and the headwater of the River Daligan. The level of total aluminium found was generally low and the pH range was not likely to promote the formation of toxic labile monomeric aluminium. The absence of significant relationships between land use and hydrochemical parameters could have been either due to the low levels of atmospheric pollution or because streams in the area were sufficiently buffered to acidification (Giller *et al.*, 1997).

National geology and soil maps are combined to produce qualitative sensitivity classes describing the relative sensitivity of surface waters to acidifying deposition (Hornung *et al.*, 1995). However, while national data of soils and geologies provide consistent information over large areas, their lack of fine detail may result in differences between the predicted water sensitivity and the results of actual field measurements. It has been found that small-scale geological anomalies, such as the localised presence of base-rich rock, are likely to reduce the sensitivity of surface waters. In contrast small areas of acid soils or geology will have little influence on the catchment with predominately carbonate rocks. Consequently, where catchments are small a national map may indicate greater sensitivity than is observed from field measurements (Hornung *et al.*, 1995; Kelly-Quinn *et al.*, 2000; Reynolds, Neal and Norris, 2001).

## 3.2 Catchment Size and Drainage

The ability of surface waters to buffer acidity has also been related to catchment size and hydrology. A number of studies in the U.K suggest that small upland catchments (<10 ha) are more susceptible to acidification than larger catchments (Neal *et al.*, 1998; Helliwell *et al.*, 2001). The reason given is that the high drainage rates and steep topography of small upland catchments reduces the contact time for runoff with bedrock and soil and consequently the time for soils to impart buffering capacity to the runoff water. As a result, waters draining smaller catchments may be more acidic and have higher concentrations of inorganic monomeric aluminium reflecting a higher proportion of runoff from the acidic mineral soils in the catchment. In larger catchments, the overall residence time of water in soil system is longer and it is therefore more effectively neutralized. However, Allot *et al.* (1990) reported that in acid-sensitive areas of the west of Ireland characterised by granite geology and blanket peat or peaty soils, catchment size may be of less significance. This is because the catchments themselves act as a source of background acidity, and groundwater is not a major component of stream flow in these areas (Allott *et al.*, 1990).

# 4 Receptor Sensitivity

## 4.1 Designation of Sensitivity

The ability of a stream or lake to neutralise acid inputs is expressed primarily by a measure of either the alkalinity or acid neutralizing capacity of the run off water. This section reviews methods for designation of sensitivity and the potential impacts on the fauna of acid-sensitive waters.

Surface waters with gran-alkalinity values of less than 10mg $l^{-1}$ as $CaCO_3$ are regarded as being acid-sensitive (Bowman, 1991). This was reflected by the Forest and Fisheries Guidelines (1991) (Anon, 1991) which set a minimum threshold of alkalinity for surface waters of 10mg $l^{-1}$ as $CaCO_3$ below which afforestation was not permitted. However, in the AQUAFOR study carried out at sites in counties Cork, Kerry Waterford and south Tipperary during 1991 and 1992, half the sites had mean alkalinity values of less than 10 mg $CaCO_3$ $l^{-1}$ yet there was no evidence of an acid-related effect on biota due to the presence of forestry (Giller *et al.*, 1997)). This led the authors to question the value of having Forest-Fisheries guidelines set at a national scale and advocate the use of a catchment based approach as local conditions in the area seemed to govern their ecology (Giller and O'Halloran, 2004). The threshold for afforestation has since been changed to 8mg $l^{-1}$ $CaCO_3$ (Forest-Service, 2004). The impact of these guidelines remains to be established. Where surface water alkalinity exceeds 15 mg $l^{-1}$ $CaCO_3$ afforestation is permitted. If the values fall in between, full, partial or no afforestation may be allowed following discussion and agreement between the Environmental Protection Agency, the Forest Service of the Department of the Marine and Natural Resources and the Regional Fisheries Board of Ireland. The response of biota, across a range of geological and soil types within this alkalinity band also has also to be assessed.

In addition, values of alkalinity depend upon flow conditions and there is often a poor relationship between peak acidity and lowest alkalinity values (Kelly-Quinn *et al.*, 1996a). In setting thresholds for surface waters it is important to consider the undesirable extremes of surface water chemistry because it is these that are most harmful to aquatic life. Recently, an index of buffering capacity the Sodium Dominance Index (SDI) has been evaluated as a suitable measure of susceptibility of streams to acidification under diverse flow conditions. While the index provided an indication of sensitivity and remained stable under conditions of variable flow the biological response along a gradient of (SDI) has to be further evaluated (Cruikshanks *et al.*, 2006).

In Wales, a mean annual hardness of 3mg $l^{-1}$ as $CaCO_3$ had been proposed as a threshold for below which afforestation was not recommended by the Welsh Water Authority (Milner, 1989). The Forestry Commission in the U.K. currently uses a catchment based critical loads assessment when designating areas of acid sensitivity.

## *4.2 Critical Loads*

In Europe, the measure used to quantify sensitivity of specific biological elements to inputs of air pollutants is the critical load (Hettelingh, Downing and de Smet, 1991; de Vries *et al.*, 1993; Hornung and Skeffington, 1993; Posch *et al.*, 1995). This is defined as 'a quantitative estimate of the exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge' (Nilsson and Grennfelt, 1988).

In Ireland, areas along the western seaboard and on the middle-east coast are considered sensitive to freshwater acidification. Critical loads have been determined for 182 lakes in these areas using the steady state water chemistry approach (SSWC) (Aherne *et al.*, 2002). This method sets the critical loads of acidity equal to the difference between the pre-industrial base cation freshwater concentration and the critical acid neutralising capacity limit ($ANC_{crit}$). The critical ANC limit is the chemical limit for protection of sensitive indicator organisms, usually fish in freshwaters. This is the most unfavourable value that the chemical criterion (in this case ANC) may attain without long-term harmful effects to ecosystem structure and function. Half the lakes surveyed (91) had a critical load of approximately 100 meq $m^{-2}$ $yr^{-1}$ or less. Twenty-four lakes (13%) had critical loads of acidity below 50 meq $m^{-2}$ $yr^{-1}$. Lakes with the lowest critical loads were located along the western seaboard, typically associated with areas considered sensitive to acidification, due to a combination of poor mineral and organic soils overlying slowly weathering base-poor geologies (Aherne *et al.*, 2002). Exceedance of critical loads gives an indication of the potential for harmful effects. Exceedance for acidity in the study was calculated as long-term deposition of non-marine (excess) sulphate plus nitrate catchment leaching (represented by freshwater concentration) minus critical loads. The critical load of acidity was exceeded for 13 lakes (7%). These were located in the uplands of Wicklow, in areas of the northwest including Donegal and along the border with Northern Ireland (Aherne *et al.*, 2002). The lakes sampled in the study were selected on the basis of their acid-sensitive soils and geology and this is generally reflected in the hydrochemical results. Thirty-seven percent of the lakes sampled had pH less than 5.5 indicating potential for severe acidification. Over 75% of the lakes had alkalinity less than 10 mg $CaCO_3$ $l^{-1}$, 33% of which had zero or negative alkalinity and consequently no buffering capacity. Of the lakes with pH<6, organic anions were the dominant or influential contributor to acidity in 65% of lakes, whereas concentrations of excess sulphate were dominant or influential in 20% (Aherne *et al.*, 2002). Most lakes, however, had relatively low levels of inorganic aluminium. It is possible that organic carbon binds much of the aluminium, which renders it relatively non-toxic to aquatic organisms. While the study detected the influence of acid deposition on hydrochemical composition, the potential impact of these changes on the ecology of lakes and their catchments remains to be established (Kelly-Quinn *et al.*, 2000)

## *4.3 Hydrochemical Impact*

There has been a considerable volume of published literature on the impacts of forest mediated acidification on the hydrochemical conditions in freshwater from several countries including Wales (Stoner *et al.*, 1984; Ormerod *et al.*, 1989; Waters and Jenkins, 1992), Scotland (Lees, 1995; Puhr *et al.*, 2000; Harriman *et al.*, 2003), Ireland (Bowman, 1991; Kelly-Quinn *et al.*, 1996a; Allott *et al.*, 1997; Giller *et al.*, 1997) and Scandinavia (Hindar *et al.*, 1995; Larssen and Holme, 2006). Some of the most relevant results from the UK and Ireland are detailed here.

Several studies in upland areas of Wales and Scotland have demonstrated that streams draining coniferous forest plantations with hard rocks and poor soils are more acidic and/or contain elevated concentrations of labile monomeric aluminium compared with those draining similar moorland catchments (Harriman and Morrison, 1982; Waters and Jenkins, 1992; Lees, 1995). These areas have historically received high pollutant inputs in the form of large volumes of moderately polluted rainfall. Furthermore, in these areas the presence of forests has been associated with the delayed hydrochemical recovery of streams as deposition of sulphate declines (Helliwell *et al.*, 2001; Harriman *et al.*, 2003). The higher acidity and in forested streams has been attributed to the enhanced capture of acidifying pollutants, (principally compounds of sulphur and nitrogen) by the forest canopy. For example, in central Scotland, a study of twelve catchments reported that mean pH values measured over one year were less than 4.4 in forested streams but ranged between 4.90 and 5.80 in moorland streams (Harriman and Morrison, 1982). In addition, total dissolved aluminium concentrations in forested streams ranged from 100 $\mu g\ l^{-1}$ to 350 $\mu g\ l^{-1}$, whereas concentrations in streams draining young forest (5 years) and moorland rarely exceeded 100 $\mu g\ l^{-1}$. The streams in the study drained areas of quartzite, schist and slate with acid peaty soils. Mean values of total hardness were less than 10 $mg\ l^{-1}$ as $CaCO_3$ (Harriman and Morrison, 1982).

While many of the studies comprised paired catchments studies, two regional scale studies by Puhr *et al.* (2000) in Scotland, and Ormerod *et al,* (1989) in Wales reported negative correlations between surface water pH and percentage catchment forest cover across a range of different catchment characteristics in acid-sensitive areas. In a survey of 113 Welsh catchments of contrasting land use, Ormerod *et al.*, (1989) reported that, within each of three different ranges of acid sensitivity (< 10, 10-15 and 15-20 mg $CaCO_3\ l^{-1}$), stream water pH declined and aluminium concentrations increased significantly with increasing percentage forest cover.

Ireland receives less atmospheric pollution than most other European countries (Aherne and Farrell, 2000). Nevertheless elevated levels of acidity and labile monomeric aluminium were recorded at forested (coniferous) acid-sensitive sites on the west coast of Ireland and in the Wicklow mountains in the east, in the period

1987- 89 (Bowman, 1991). In contrast, a study of forested sites in south Munster (in counties Cork, Waterford and Kerry) on Old Red sandstone found no evidence of forest-mediated acidification (Giller *et al.*, 1997).

Acidity in both the Wicklow and Galway-Mayo studies was essentially episodic in nature. The sites examined were in areas of base poor geologies with acid-sensitive or acidic soils. Alkalinity at these sites was generally less than 10 mg $CaCO_3$ $l^{-1}$ during periods of low flow and fell to zero or became negative during high flow (Kelly-Quinn *et al.*, 1996a). Continuous monitoring of high flow events at certain streams found that acid episodes at some forested sites were more severe and more prolonged than at moorland sites. These episodes tended to occur during periods of high flow in winter and spring. It was also found that changes in pH, occurred more rapidly at those forested streams monitored (Kelly-Quinn *et al.*, 1996a; Allott *et al.*, 1997). Furthermore, elevated concentrations of labile monomeric aluminium (>50 $\mu gl^{-1)}$ were associated with streams which had a high cover of mature plantation forestry (>25%) and where dissolved organic carbon (DTOC) concentrations were low (< 7.0 mg $l^{-1)}$ (Kelly-Quinn *et al.*, 1996a)

The sites examined in Galway and south -Mayo were located in areas of granite, quartzite, schist and gneiss, and the soils at these sites were predominantly blanket peats. Alkalinity values measured over two winters (November - March, 1990/91 and October – April 1991/92) during periods of high flow were low, particularly in sites in south Galway on granite, where mean alkalinities were either close to zero or negative. A positive correlation was found between $H^+$ and percentage catchment forest cover across all sites ($R^2 = 0.674$) (Allott *et al.*, 1997). Trends in concentrations of labile monomeric aluminium were coincident with those of $H^+$. The greatest concentrations occurred at sites with the highest proportion of closed canopy forestry. During the first winter of the study, concentrations of labile monomeric aluminium at forested sites (>25% catchment cover) were in the range of 54 to 130 $\mu g\ l^{-1}$, with a maximum of over 200 $\mu g\ l^{-1}$ (Allott *et al.*, 1997).

Of the sites examined by the AQUAFOR study in south Munster, set largely on Old Red Sandstone, there was no obvious relationship between forest cover and stream acidity. In 42% of the sites, spring alkalinity values were less than 10 mg $CaCO_3$ $l^{-1}$ in 1992, yet as previously mentioned there was no evidence of acidification at forested sites. Only three sites could be considered acidic (i.e. the pH value approached 5): two headwaters of the Araglin river and the headwater of the River Daligan (Giller *et al.*, 1997).

## *4.4 Impacts on Aquatic Biota*

Several studies in poorly buffered areas have found aquatic biota to be impacted as a result of the influence of forests on surface water chemistry (Harriman and Morrison, 1982; Stoner *et al.*, 1984; Ormerod *et al.*, 1993; Allott *et al.*, 1997; Tierney, Kelly-Quinn and Bracken, 1998). In almost all cases, this has been attributed to the role of mature coniferous forest plantations in intercepting atmospheric pollutants, which resulted in increased acidity and/or concentrations of labile monomeric aluminium in these streams.

The ecological effects of acidification may act directly though toxicity to biota resulting in increased mortality and/or physiological stress or indirectly through ecosystem processes such as energy transfer (Ormerod *et al.*, 1991b). Toxicity arises due to acidity or the presence of labile monomeric aluminium. The parameters of importance in relation to biotic impacts include pH, inorganic aluminium and calcium (Ormerod and Wade, 1990). Calcium is important both in terms of conveying buffering capacity and in ameliorating the effects of acid stress and aluminium toxicity on fish (Brown, 1983), if $Ca^{2+}$ values are in excess of 1.0 mg $l^{-1}$. Biological impacts associated with acidification in streams include reductions in or total elimination of fish populations, the elimination of some acid-sensitive invertebrate groups and changes in the quality of primary producers (Stoner *et al.*, 1984; Ormerod, Wade and Gee, 1987; Rees and Ribbens, 1995; Tierney *et al.*, 1998).

### 4.4.1 Fish

Upland streams are important nursery areas for salmonids in Ireland. Acidification, accompanied by high concentrations of $H^+$ and elevated levels of labile monomeric aluminium, impoverishes fish communities (Driscoll *et al.*, 1980). Responses vary, however, from species to species and between developmental stages, and even among different populations of the same species (Rosseland, 1986; Rosseland and Skogheim, 1987). Underyearlings are generally more vulnerable to acidification than adults. The egg and alevin stages are considered to be the most sensitive (Sayer, Reader and Dalziel, 1993). Consequently, low pH at the time of spawning and during the early stages of development is critical to the survival of fish. The initial reaction of acid stressed fish is to migrate to reaches with less acidity (Baker *et al.*, 1996). Low stream pH has also been found to affect migratory behaviour. In weakly acidic environments salmonids have also been found to avoid acidic water in selection of spawning sites (Ikuta *et al.*, 2001; Ikuta, Suzuki and Kitamura, 2003). When this migration is not achieved, either through lack of ability or extensive areas of poor water quality, spawning may cease and eggs may not be laid or hatched leading to recruitment failure (Ikuta *et al.*, 2003). Hatching in low pH waters may be impaired by a lack of development resulting in the inability to escape from the egg membrane (Sayer *et al.*, 1993). Ultimately the population is eliminated (Beamish, 1976). While mortality is most prevalent among the early sensitive stages of fish development, adult fish death has been reported associated with episodes of sudden acidification of water. This phenomenon usually coincides with increased runoff due to snow melt in the spring or heavy autumn rain (Hesthagen, 1989; Reader and Dempsey, 1989). Fish mortality is largely due to the physiological effect of hydrogen ion and/or aluminium stress. Exposure to acid stress alone does not cause physiological damage until the pH drops below 5.0 (Wood, 1989). The combination of acid/inorganic monomeric aluminium stress is more complex. At low pH levels (~ 4.0) aluminium may have a beneficial effect by reducing the impact of hydrogen ion stress (Baker and Schofield, 1982). Inorganic aluminium is considered to be most toxic to fish at a pH range of between 5.0 and 5.5 (Brown and Sadler, 1989). These chemical conditions together with low concentrations of calcium have been found to interfere with physiological mechanisms in adult fish e.g. regulation of ion exchange across gill membranes (Howells *et al.*, 1990; Havas and Rosseland, 1995) leading to physiological stress and death (Schofield, 1976; Reader and Dempsey, 1989).

The absence of trout in forest streams with high concentrations of aluminium has been reported from studies in Scotland (Harriman and Morrison, 1982; Rees and Ribbens, 1995) and acid sensitive streams in Wicklow, in Ireland (Bowman, 1991; Kelly-Quinn *et al.*, 1996b). Stoner *et al.* (1984) found that in acid-sensitive streams in mid-Wales, trout (*Salmo trutta* L.) were absent from conifer-forested streams with average total hardness < 10mg $CaCO_3$ $l^{-1}$, but were present in unforested streams with the same hardness. The average value of pH in these streams was less than 5.5 and concentrations of dissolved inorganic aluminium frequently exceeded 50 µg $l^{-1}$. In Galloway in Scotland, Rees and Ribbens (1995) reported a clear relationship between fish populations and water chemistry; numbers of trout were positively correlated with pH ($r^2$=0.69; p<0.001), and with concentrations of calcium ($r^2$=0.55; p<0.001).

In Ireland, the influence of forestry on fish populations as mediated through stream water chemistry was examined by the AQUAFOR studies. Allott *et al.* (1997) reported that salmonid stock densities were lower at forested than unforested sites in eight streams draining granite in Galway and absent from those sites with a minimum pH value of less than 4.2. In Wicklow, trout density was found to be positively associated with alkalinity and calcium and negatively correlated with hydrogen ion concentration. Four sites on three streams (Annalecka, Vartry and Lugduff) had good trout habitat but were fishless. These sites had a high % (>25%) mature forest cover in their catchments (Kelly-Quinn *et al.*, 1996b). Stream chemistry at the sites was characterised by consistently high levels of acidity and concentrations of inorganic aluminium greater than 40µg $l^{-1}$, the threshold considered toxic to salmonids (Turnpenny *et al.*, 1987). This hypothesis was based on the results of experimental work in the Lugduff River, where high levels of mortality among caged brown trout were attributed to underlying base poor geology together with anthropogenically derived acidification, exacerbated by afforestation, which resulted in the extensive coating of aluminium and mucus on the fish gills. No evidence of stress was observed in the control, non-forested stream (Bowman and Bracken, 1993).

The relationship between forestry and fish populations was examined at sites in Cork as part of the Munster AQUAFOR study. Preliminary analysis suggested that salmonid populations at sites with low to medium afforestation levels were larger than corresponding sites with zero afforestation, but at high afforestation levels, fish stocks were usually lowest. However, it is unlikely that this decrease was related to acidification related factors (Giller *et al.*, 1997). These findings are supported by a subsequent, more extensive study of 36 streams in counties Cork and Kerry in Munster, which showed that while the lowest salmonid populations were found at sites of highest forest cover, this was not always the case and so overall no negative effects of forest cover were found on trout were attributed to the presence of coniferous forest (Lehane *et al.*, 2004).

Although almost all studies have examined the ecological effects of acidic pollutants, namely sulphur and nitrogen, in some cases sea-salt episodes have led to salmonid mortalities (Hindar *et al.*, 1994; Barlaup and Aatland, 1996; Teien *et al.*, 2004b). In western Norway, a major storm event in January 1993 led to toxic conditions for fish in two coastal watersheds (Barlaup and Aatland, 1996). Values of pH dropped to 4.5-5.1 and concentrations of labile aluminium were high (200 – 300 $\mu g\ l^{-1}$) during this event. These conditions led to episodic mortalities of brown trout (*Salmo trutta*). However, no major impact on egg survival or later recruitment of fry was recorded (Barlaup and Aatland, 1996).

### 4.4.2 Benthic Macroinvertebrates

Stony streams normally support a characteristic fauna of benthic macroinvertebrates, predominantly insects such as Ephemeroptera (mayfly), Plecoptera (stonefly), Trichoptera (caddis-fly) and Diptera (true-flies). Also present are flatworms, oligochaetes and a few crustaceans and molluscs (Sutcliffe and Hildrew, 1989). The composition of the macroinvertebrate community reflects the prevailing conditions over time and so finds widespread use as an index of water quality (see (Wright, 1995)). In anthropogenically-acidified streams where the pH is below 5.7-5.4 there is a distinct change in the benthic invertebrate faunal community composition. Some taxa disappear or become scarce, particularly mayflies, some caddisflies, crustaceans such as *Gammarus* sp. and molluscs. As a result, species diversity is reduced (Sutcliffe and Hildrew, 1989; Wade, Ormerod and Gee, 1989; Rosemond *et al.*, 1992; Ventura and Harper, 1996; Guerold *et al.*, 2000). These taxa are impacted directly by low pH and high inorganic aluminium concentrations (Fjellheim and Raddum, 1990). The surviving communities are typically dominated by nemourid stonefly species (and include *Leuctra, Protonemura and Nemoura*) (Ventura and Harper, 1996; Friberg, Rebsdorf and Larsen, 1998; Dangles and Guérold, 2000; Thomsen and Friberg, 2002). These taxa are considered more acid tolerant and may benefit from the absence of specialist feeders by opportunistic feeding behaviour (Ledger and Hildrew, 2000).

In contrast, studies of naturally acidic streams suggest that they support a diverse and functional macroinvertebrate community (Collier *et al.*, 1990; Dangles, Malmqvist and Laudon, 2004b). Naturally acidic streams often have high levels of dissolved humic substances, which may ameliorate the toxicity of metals, primarily inorganic aluminium, which has documented toxic effects on biota in acidified freshwaters (Driscoll *et al.,* 1980). Where low pH is brought about naturally by high concentration of organic acids, most dissolved aluminium is rendered non-toxic through complexation with dissolved organic matter (Kullberg, 1992; Dangles *et al.*, 2004b). A second reason proposed to account for the diversity of macroinvertebrates in naturally acidic streams is that fish are sensitive to low pH and tend to be present in reduced numbers and as a result the invertebrates are subject to lower predation pressure (Dangles *et al.,* 2004).

In areas of atmospheric pollutant deposition, forested streams draining acid-sensitive areas have been found to be more acidic and/or have higher concentrations of labile monomeric aluminium when compared to the communities occurring in unforested sites. This is reflected in the invertebrate community, which support fewer taxa, fewer individuals and a different assemblage of species than unforested sites (Harriman and Morrison, 1982; Ormerod *et al.*, 1993; Friberg *et al.*, 1998; Ormerod *et al.*, 2004).

The study referred to above by Stoner *et al.* (1984) also examined the hydrochemistry and invertebrate fauna of 13 streams draining areas of Ordovician and Silurian sedimentary rock with oligomorphic peat, ferric stagnopodzols and brown podzolic soils. It was reported that the forested streams had a more restricted invertebrate community (23 -37 taxa) than the non-forested streams (46-78taxa). Species of Ephemeroptera, (*Baetis rhodani* (Pict.), *Rhithrogena semicolorata* (Curtis) and *Electrogena lateralis* (Curtis)) were numerically important (up to 46%) at most moorland sites but were scarce (<1%) or absent under forest. Similarly, the number of trichopteran (caddisfly) species was greater in moorland streams (9-17 taxa) than coniferous-forested streams (4-8). Plecoptera were most abundant at forested sites, comprising 50 to 75% of individuals. The differences between the communities were attributed to the episodic deposition of acidity arising from the increased deposition of excess (non-marine) sulphate in the forested catchments (Stoner *et al.*, 1984). Furthermore, a survey of 66 predominantly upland streams (1$^{st}$ -3$^{rd}$ order) throughout Wales and Scotland reported that for any given pH, aluminium concentrations (fraction not determined) were significantly higher in streams draining conifer catchments than streams draining whole catchments of moorland or deciduous woodland. In addition, taxon richness of Ephemeroptera, Plecoptera, Trichoptera and all taxa combined decreased significantly with increasing acidity and aluminium concentration (Ormerod *et al.*, 1993).

In Ireland, the effects of forestry on macroinvertebrates has been assessed by biannual surveys of acid-sensitive lakes and their inflow streams by the Environmental Protection Agency, as well as by the AQUAFOR studies of the early 1990's. The effect of forestry varies geographically and is related to the underlying geology and the forestry influence on water chemistry.

In Wicklow, a survey of 47 upland streams showed that those flowing over granite supported a less diverse fauna than those flowing over Palaeozoic sedimentary beds. However, within each geological type non-forested streams supported a more diverse fauna than adjacent forested sites (Tierney *et al.*, 1998). Species of Ephemeroptera (*B. rhodani, Siphlonurus lacustris* (Eaton) and *Ameletus inopinatus* (Eaton)) were absent from 8 sites, which had high percentage forest cover (>25% with the exception of Lugduff 15%-25%) (Tierney *et al.*, 1998). The impacts on the benthic macroinvertebrate community found at the forested sites were attributed to episodically low pH and elevated concentrations of labile monomeric aluminium conditions (Kelly-Quinn *et al.*, 1996a).

Long term monitoring of the forested Lugduff stream, an inflow stream of Glenadalough Lake Upper in Wicklow by the Environmental Protection Agency shows that this stream continues to be impacted by anthropogenic acidification (Toner *et al.*, 2005). Initial, chemical and biological data from the period 1987-1989 showed that the stream had consistently high levels of labile aluminium (median concentrations 194 µq $l^{-1}$) and low pH and in consequence a very restricted fauna. An adjoining stream from a non-forested catchment (Glenealo River) did not show evidence of being adversely affected by acidification and an abundant and diverse fauna, including the acid-sensitive genus *Baetis* in particular *B. rhodani* was present on all occasions examined. In contrast, the absence of Ephemeroptera and other acid-sensitive organisms was a significant feature of a restricted fauna, dominated by the order Plecoptera at the Lugduff River (Bowman, 1991). Subsequent biannual monitoring of these sites shows that the Lugduff stream and the littoral fauna of the lake continue to be impacted. Acid-sensitive macroinvertebrate fauna are absent from these sites but present in the non-forested Glenealo river (Toner *et al.*, 2005).

The influence of forestry on the benthic macroinvertebrate community was also examined at ten sites (5 forested and 5 unforested) in poorly buffered areas of Galway and south Mayo (Allott *et al.*, 1997). Of the ten sites examined, there was a trend towards lower numbers of invertebrate taxa at forested sites. However, the invertebrate community in these streams was also related to the underlying geology. Crustaceans such as *Gammarus* and molluscs were absent from all but one site, regardless of forest cover. Furthermore, at sites located in granite areas of Galway, only acid tolerant Ephemeroptera, *Leptophlebia vespertina* L. were present, whereas sites in the Inagh Valley on quartz and schist supported a more diverse Ephemeropteran community. Similarly, biannual surveys of the acid-sensitive Lough Maumwee catchment between 1984 and 2000, have recorded 15 Ephemeroptera taxa (Jim Bowman pers comm.). The catchment of Laugh Maumwee consists of a quartzite formation while the geology immediately around the lake is granite (Allott *et al.*, 1997). Thus, the number of Ephemeropteran species encountered appears to be highly variable in this region and the influence of forestry and geology requires further clarification.

As part of the AQUAFOR study in Munster, similar biological surveys were carried out on 16 sites in generally less acid-sensitive areas of Cork from 1990 to 1992 (Giller *et al.*, 1997). Here study the influence of land use on macroinvertebrate communities at sites in Cork was secondary to that of altitude. Some macroinvertebrate communities at medium altitude (200-300m) with medium to high levels of forestry (25 to> 50%) seemed to resemble communities at higher altitudes (>300m) than sites with low levels of forest cover at a medium altitude. Changes to the macroinvertebrate communities at these sites were considered to be more likely the result of physical rather than chemical factors (Giller *et al.*, 1997). These findings were supported by further work in Kilworth forest (Douglas River) where macroinvertebrate communities within forest cover were not found to be impoverished (Clenaghan *et al.*, 1998).

The adverse impacts on the macroinvertebrate community in the streams in Wicklow were attributed to the occurrence of acid episodes. The impact of acid episodes on the benthic macroinvertebrate communities of acid streams has been highlighted by a number of other studies (Hall, 1994; Kratz, Cooper and Melack, 1994; Lepori, Barbieri and Ormerod, 2003; Lepori and Ormerod, 2005). For example, Kowalik and Ormerod (2006) found that Ephmeroptera were absent from acidic streams and were restricted to one species (*B. rhodani*) in episodically acidic streams in Wales. The populations of *B. rhodani* in the episodic streams were transient, occurring in summer and disappearing in autumn and winter when pH levels dropped to 5.0. It is likely that over wintering nymphs experienced high mortality due to acid episodes in combination with high concentrations of inorganic aluminium (Kowalik and Ormerod, 2006). This hypothesis was supported by transplantation experiments carried out as part of the same study which found high mortality of nymphs of *B. rhodani,* when exposed to acidic conditions (pH values 3.8-4.5) for periods of 4 days at a time (Kowalik and Ormerod, 2006).

While the impacts of forests on the macroinvertebrate community at the sites in Wicklow were primarily related to stream chemistry, it is also likely that some of the differences between control and forested sites were due to the physical and energetic impacts of coniferous forest in the riparian zone. Coniferous forest in influences habitat structure (Ormerod et al., 1993), food quality (Thomsen and Friberg, 2002) and light inputs (Melody and John, 2004). Coniferous needles have lower food quality than broadleaves (Friberg and Jacobsen, 1999) and this coupled with the loss of efficient shredders such as Gammarus species due to acid toxicity (Dangles et al., 2004a), may result in reduced rates of leaf breakdown and consequently limit secondary productivity (Thomsen and Friberg, 2002). It should be noted here that it has been standard practice in Ireland to leave an unplanted buffer strip beside the aquatic zone since the mid-1980s both at afforestation and reforestation stages.

While long-term monitoring data are available for acid-sensitive lakes including the forested Lugduff stream in Wicklow, there is a lack of long-term data for other sites. This is important to take into account the response of the invertebrate communities to inter-annual variation in precipitation and pollutant deposition. For example, work carried out in the Annalecka stream in 2003, in Wicklow that was considered to be impacted previously showed a partial recovery of the macroinvertebrate community with good numbers of the mayfly *B. rhodani* present. This may have been due to a reduction in forest cover arising from the creation of a buffer zone along the stream following harvesting of the adjacent forest combined with low seasonal rainfall for a number of preceding years (Cruikshanks *et al.*, in press).

### 4.4.3 Primary Production

Biological elements used to determine ‘good ecological status’ also include phytoplankton, macrophytes and phytobenthos. Most acidified areas lie within landscapes that have hard inert bedrock; the rivers in such landscapes are generally fast flowing and the main vegetation types are phytobenthos and bryophytes. The current velocity is usually too high to develop a significant biomass of phytoplankton. Several changes appear during acidification, which are reported to affect structural communities of primary producers in rivers (Niyogi, McKnight and Lewis, 1999; Thiébaut and Muller, 1999). For example, Ormerod, Wade and Gee (1987) found that macro-floral assemblages were related most strongly to pH and aluminium concentration. *Scapania undulata* (L.), *Nardia compressa* ((Hook.) Gray.) and filamentous chlorophytes characterized streams of mean pH 5.2-5.8, whereas *Fontinalis squamosa* (Hedw.) and *Lemanea* species occurred at higher pH values (5.6-6.2 and 5.8-7.0, respectively). Furthermore, *F. squamosa* and *Lemanea* spp. were negatively associated with forest cover (>50%) (Ormerod, Wade and Gee, 1987). Diatoms may also be used to provide an indication of short-term acidification events in streams. Hirst *et al.* (2004), examined the response of biofilm to periods of acid conditions using transplantation experiments and diffusing substrates in streams in Llyn Brianne, Wales (Hirst *et al.*, 2004). While studies such as these suggest an effect through altered stream chemistry, separate work will be necessary in Ireland to establish the impacts of acidity on these biological elements.

### 4.4.4 Birds

The riverine dipper, *Cinclus cinclus* (L.) is a bird closely dependent on benthic invertebrates as a food source. Although not one of the ecological quality elements included in the Water Framework Directive, its status as a high ranking predator integrates conditions across trophic levels and so acts as a valuable indicator of stream biological quality (Buckton *et al.*, 1998). The distribution of dippers has been surveyed along a number of sites on soft water streams, throughout Wales. Sites without dippers had lower pH and/or high concentrations of inorganic aluminium, fewer trichopteran larvae and Ephemeropteran nymphs, and were on rivers with more conifer forest cover on their catchments, than sites where dippers were present (Ormerod *et al.*, 1986; Buckton *et al.*, 1998). Furthermore, breeding biology parameters (e.g. date of first egg, brood size at hatching and fledgling) were adversely impacted on acidic streams in Wales, compared with circum-neutral streams (Ormerod *et al.*, 1991a). In Ireland, the presence of dippers was surveyed in the Araglin Valley, Cork as part of the Munster AQUAFOR study over two breeding seasons, 1991-1992. Here dippers were located at sites irrespective of land use and there were no differences in breeding biology parameters across sites. In fact, the density of dipper nests appeared to be highest in the forested sub catchment (Giller *et al.*, 1997). These results are supported by other studies which have found no negative impacts of forest cover on dipper populations or breeding biology in south west Ireland (Smiddy *et al.*, 1995).

# 5 Measures

Reduction in the emission of acid pollutants is the only way of solving the general problem of surface water acidification. Emissions agreements such as the Gothenburg Protocol (1999) and the Large Combustion Plant Directive (DIRECTIVE 2001/80/EC) have set emission ceilings for pollutants such as sulphur, $NO_x$ and ammonia. However, the increased capture of acidic pollutants by forests poses a risk of delaying the recovery of acidified waters and acidification may still continue to occur in the most sensitive areas for some time. However, some measures can be implemented to minimize the impact of forest-mediated acidification. An extensive range of measures have been developed by the Forest Service of the Department of Agriculture and Food which respond to the findings of studies carried out in Ireland such as the AQUAFOR studies mentioned above. Other measures such as the use of liming and buffer strips to mitigate acidification are discussed briefly below.

## *5.1 Forest Guidelines*

It is important to note that the nature of forestry practices in Ireland have changed dramatically since the forests were initially planted in the 1950s-1970s.. In response to the findings of international studies and in particular those conducted in the UK, which have been detailed in this review, the Fisheries & Forestry Service (of the time) circulated to local forest management in the early 1980s water protection measures related to forest operations with the explicit aim of minimising or preventing an impact from forests and forest operations on surface waters. This was followed up by the newly formed Forest Service in 1991 with the issuing of the more comprehensive Forestry and Fisheries Guidelines. Following the publication of the AQUAFOR report, these Guidelines were further strengthened and augmented by the publication of the Code of Best Practice (Forest-Service, 2000), Forestry and Water Quality Guidelines (Anon, 2000a), Forest Harvesting and Environment Guidelines (Anon, 2000b), current Afforestation Grant and Premium Schemes Manual (Forest

Service, 2004) and the introduction of statutory regulations on aerial fertilisation (Anon, 2006). While it is not the purpose of this review to review current forestry practices, a few of the measures relevant to the topics discussed above are outlined here. For example, since the introduction of the Forest and Fisheries Guidelines (1991) all plantations require the establishment of buffer zones adjacent to aquatic zones (where an aquatic zone is defined as a permanent or seasonal river or lake shown on an Ordnance Survey 6 inch map). A number of guidelines are also set out in terms of ground preparation. The guidelines require the installation of drains and sediment traps designed to minimise flow velocities. For example, drains are excavated at an acute angle to the contour gradient to minimise flow velocities and taper out before entering the buffer zone and so do not discharge directly to the aquatic zone (Anon, 2000a). In relation to felling, current forest management practices involve a move away from even aged stands to mixed age stands with phased felling. Furthermore, a number of factors such as soil type, slope and the sensitivity of the aquatic zone are taken into account in determining coupe size (Anon, 2000b)

As well as the introduction of Forest and Water Quality Guidelines, the species of trees planted and the nature of areas being afforested have changed. Traditionally forestry was restricted to low-productivity acid soils unsuitable for agriculture. However, under the current Afforestation Grant and Premium Scheme (2004), the majority of planting is taking place on better quality land ('enclosed and improved land'). Furthermore, the majority of new forestry being planted is by private operators. In 2005, virtually all new planting was carried out by private forestry 57% of which was coniferous and 26% broadleaf. This is in contrast to 30 years ago when almost all afforestation was undertaken by the State and comprised predominantly coniferous species (source Forest Service). The cumulative effects of crop restructuring, smaller coup size, the introduction of buffer zones and open spaces and replanting with a more diverse tree species range are likely to have a significant bearing on the future potential acidification pressure arising from forests on surface waters.

In terms of addressing the potential acidification pressure arising from the scavenging effect of trees, the Forest Service of the Dept of Agriculture and Food uses alkalinity readings in decision-making relating to afforestation grants for potentially acid-sensitive catchments. As mentioned previously, where the minimum alkalinity of runoff water is less than 8mg $l^{-1}$ $CaCO_3$ no afforestation is permitted. In areas where alkalinity exceeds 15 mg $l^{-1}$ $CaCO_3$ afforestation is permitted. If the values fall in between, full, partial or no afforestation may be allowed following discussion and agreement between the Environmental Protection Agency, the Forest Service of the Department of the Marine and Natural Resources and the Regional Fisheries Board. Sampling must take place on a minimum of four occasions between 1st February and 31st May at intervals of not greater than 4 weeks (Forest Service, 2004).

## *5.2 Liming*

Symptomatic treatment of anthropogenically driven surface-water acidification involves the addition of neutralising agents such as powdered limestone to affected water bodies or their catchments has been extensively used in Europe. The application of lime generally results in an increase in pH, calcium and alkalinity and a reduction in the concentration of dissolved aluminium and other metals (Weatherley, 1988; Degerman and Appelberg, 1992; Lingdell and Engblom, 1995; McKie, Petrin and Malmqvist, 2006). However, biological responses to liming are varied. In some cases, liming provides adequate chemical conditions for the recovery of acid-sensitive macroinvertebrates (Fjellheim and Raddum, 1995; Lingdell and Engblom, 1995; Clayton and Menendez, 1996; Raddum and Fjellheim, 2003). However, biological responses on other occasions have been less successful (Simmons and Doyle, 1996; LeFevre and Sharpe, 2002; Keener and Sharpe, 2005). Bradley and Ormerod, (2002) examined the response of the macroinvertebrate community over ten years in streams treated with a single liming application. As a result of this one treatment, mean pH remained above 6, calcium concentration increased (>2.5 mg $l^{-1}$) and concentrations of aluminium remained low (< 0.1mg $l^{-1}$) for the following ten years. Abundance of acid-sensitive taxa increased significantly, but only during the first two years following treatment. Furthermore, on average, only 2-3 acid-sensitive taxa were added to the treated streams, roughly one-third their average richness in adjacent circumneutral streams (Bradley and Ormerod, 2002). One possible reason for the lack of recovery is that acid episodes continue to affect the stream fauna. The occurrence of acid episodes in treated water bodies has been found to impair the recovery of acid-sensitive macroinvertebrates and fish (Degerman and Appelberg, 1992; Fjellheim and Raddum, 1995; Bradley and Ormerod, 2002).

The application of lime to acidified surface waters has not taken place in Ireland. A review of liming options concluded that the most promising method appropriate for Ireland was the stream dosing method. However, it was concluded that a better understanding of the hydrology of small acid steams, particularly those draining peaty catchments is required before a complete assessment of the effectiveness of catchment liming can be made (Donnelly, Jennings and Allott, 2003).

## 5.3 Buffer zones to mitigate acid runoff

The effectiveness of buffer zones to mitigate acid water draining forested catchments has been assessed by a limited number of studies, the results of which suggest that buffer strips are not effective in buffering acid runoff (Weston, 1995; Cowan, 1998; Jennings, Donnelly and Allott, 2002). Examples from mid-Wales show that bank-side liming in forest streams have had no measurable effect on stream chemistry. It is likely that riparian buffer strips were breached by soil pipes, drainage channels and over land flow so that chemical effects from the major part of the catchment still dominated stream chemistry (Hornung, Brown and Ranson, 1990a; Ormerod *et al.*, 1993).

# 6 Knowledge Gaps

There exist a number of significant knowledge gaps relating to the interaction between forestry and surface water acidification in Ireland. These are discussed under the categories 'pressure', 'pathway' and 'receptor' below.

## 6.1 The Pressure

The deposition load arising due to the scavenging effect of coniferous forests depends on a number of factors including inter alia, the prevailing chemical climate, site aspect, slope, altitude as well as precipitation and wind direction. However, the impact of forests in altering the load of atmospheric inputs to the catchment has not been modelled for Irish conditions. There are a number of knowledge gaps relating to this such as what contribution occult deposition makes to the deposition load of Irish forests. Secondly, where impacts on stream biota have been deemed to occur, it has not been established as to the amount of upstream forest cover which gives rise to an adverse impact. There is also a lack of long-term data monitoring the influence of forests on surface water chemistry and biota in many acid-sensitive areas in Ireland. This is significant, as stream hydrochemistry has been found to vary from year to year with precipitation levels, frequency of storm events and the direction of prevailing air masses. In addition as mentioned above, the nature of forest practices, species mix and the deposition of sulphur and nitrogen has changed considerably over the past twenty years or more. As such, it is relevant to examine what impact these changes will have on the interaction between plantation forests and surface water chemistry for acid sensitive waters.

## 6.2 Pathway Susceptibility

The varying results of the biological surveys of the AQUAFOR study in counties Wicklow, Galway-Mayo and Cork, Kerry and Waterford in Munster point to the influence of regional variations in determining pathway susceptibility. The current Forest Service guidelines (Forest Service, 2004) stipulate that below a minimum alkalinity of less than 8mg $l^{-1}$ $CaCO_3$ no afforestation is permitted. However, the impact of these guidelines on aquatic biota has yet to be determined across the different acid-sensitive geologies and soil types in Ireland. Equally, the response of aquatic biota to afforestation within the band of 8 – 15 mg $l^{-1}$ $CaCO_3$ has not been established. It has not been established that the measurement of alkalinity is the most appropriate measure of acidy in preference to other methods such as Sodium Dominance Index or Acid Neutralising Capacity. This is a topic that cannot be overlooked or ignored, especially after the findings of the AQUAFOR report on this matter.

Other factors which may influence the pathway susceptibility but for which little is known include, the effect of lakes upstream of a forested catchment and the presence of mixed geologies and/or soil types, both of which may be sufficient to significantly alter the hydrochemistry of associated surface waters (Hornung *et al.*, 1995).

## *6.3 Receptor Sensitivity*

There are also a number of significant knowledge gaps in relation to the impact of forest-mediated acidification on aquatic biota in acid-sensitive areas. Firstly, it is necessary to establish the extent of forest-related impacts on various biotic components. While a long-term record exists for the forested Lugduff stream and Glendalough Lake Upper in County Wicklow, long term monitoring data for other forested sites in acid-sensitive areas does not exist. The AQUAFOR studies that examined the interaction between plantation forests and invertebrate and fish communities are now 15 years old. As such current knowledge of the influence of plantation forests on aquatic biota in acid sensitive areas needs to be updated. In addition, the existence of an impact, if any, arising from forest plantations in areas of high background natural acidity in areas of the west of Ireland remains to be established.

Furthermore, biological elements used to determine ‘good ecological status’, under the Water Framework Directive include phytoplankton, macrophytes and phytobenthos. There is a lack of data surrounding the response of these elements to pressures such as forestry, in Ireland.

# 7 References


Aber, J.D., Nadelhoffer, K.J., Steudler, P., and Melillo, J.M. 1989 Nitrogen Saturation in Northern Forest Ecosystems. *BioScience*, 39(6): 378-386.

Aherne, J., and Farrell, E.P. (2000) Final Report: determination and mapping of critical loads for sulphur and nitrogen and critical levels for ozone in Ireland. Dublin: Environmental Protection Agency, pp.212.

Aherne, J., Kelly-Quinn, M., and Farrell, E.P. 2002 A Survey of Lakes in the Republic of Ireland: Hydrochemical Characteristics and Acid Sensitivity. *AMBIO: A Journal of the Human Environment*, 31(6): 452-459.

Alexander, C.E., and Cresser, M.S. 1995 An assessment of the possible impact of expansion of native woodland cover on the chemistry of Scottish freshwaters. *Forest Ecology and Management*, 73(1-3): 1-27.

Allott, N., Brennan, M., Cooke, D., Reynolds, J., and Simon, N. (1997) AQUAFOR Report 4 Stream chemistry, hydrology and biota Galway-Mayo region. In A study of the effects of stream hydrology and water chemistry in forested catchments on fish and macroinvertebrates. 4), Dublin, Ireland: COFORD.

Allott, N., Mills, W.R.P., Dick, J.R.W., Eacrett, A.M., Brennan, M.T., Clandillon, S., Phillips, W.E.A., Critchley, M., and Mullins, T.E. (1990) Acidification of Surface Waters in Connemara and South Mayo. Current Status and Causes, Dublin: duQuesne limited, Economic and Environmental Consultants, pp 61.

Anon. (1991) Forest and Fisheries Guidelines, Dublin, Ireland: Forest Service Department of Agriculture and Food.

Anon. (2000a) Forest and Water Quality Guidelines, Dublin, Ireland: Forest Service of the Department of Agriculture and Food, pp 12.

Anon. (2000b) Forest Harvesting and the Envirionment Guidelines, Dublin, Ireland: Forest Service of the Department of Agriculture and Food, pp 16.

Anon. (2004) Guidance on thresholds and methodology to be applied in Ireland's River Basin Districts. Surface Water Guidance Document. Paper by the Working Group on Characterisation and Risk Assessment. .

Anon. (2005) The characterisation and analysis of Ireland's river basin districts in accordance with Section 7(2 & 3) of the European Communities (Water Policy) Regulations 2003 (SI 722 of 2003). National Summary Report (Ireland).

Anon. (2006) S.I. No. 592 of 2006 European Communities (Aerial Fertilisation) (Forestry) Regulations 2006

Baker, J.P., and Schofield, C.L. 1982 Aluminum toxicity to fish in acidic waters. *Water, Air, & Soil Pollution*, 18(1): 289-309.

Baker, J.P., Sickle, J.V., Gagen, C.J., DeWalle, D.R., Sharpe, W.E., Carline, R.F., Baldigo, B.P., Murdoch, P.S., Bath, D.W., a. Krester, W., a. Simonin, H., and Wigington, P.J., Jr. 1996 Episodic Acidification of Small Streams in the Northeastern United States: Effects on Fish Populations. *Ecological Applications*, 6(2): 422-437.

Barlaup, B.T., and Aatland, A. 1996 Episodic mortality of brown trout (Salmo trutta L.) caused by sea-salt-induced acidification in western Norway: Effects on different life stages within three populations. *Canadian Journal of Fisheries and Aquatic Sciences*, 53(8): 1835-1843.

Bashir, W., McGovern, F., Ryan, M., Burke, L., and Paull, B. (2006a) Long-Term Trends in Atmospheric Pollutants at Valentia Observatory, Ireland.

Environmental Research Centre Report Johnstown Castle, Co. Wexford, Ireland: Environmental Protection Agency pp 11.
Bashir, W., Ryan, M., Burke, L., McGovern, F., and Paull, B. (2006b) An analysis of the ionic composition of Irish precipitation and background air quality since 1980 based on samples collected at Valentia Observatory, Co. Kerry, Ireland. In Air Pollution 2006 (J.W.S. LONGHURST, and C.A. BREBBIA, eds): WIT Press.
Beamish, R.J. 1976 Acidification of lakes in Canada by acid precipitation and the resulting effects on fishes. *Water, Air, & Soil Pollution*, 6(2): 501-514.
Bowman, J. (1991) Acid sensitive surface waters in Ireland, Dublin: Environmental Research Unit, pp 321.
Bowman, J., and Bracken, J. 1993 Effects of run-off from afforested and non-afforested catchments on the survival of brown trout Salmo trutta L. in two acid-sensitive rivers in Wicklow, Ireland. *Biology and Environment: Proceedings of the Royal Irish Academy*, 93(B): 143-150.
Bradley, D.C., and Ormerod, S.J. 2002 Long-term effects of catchment liming on invertebrates in upland streams. *Freshwater Biology*, 47: 161-171.
Bredemeier, M., Blanck, K., Xu, Y.J., Tietema, A., Boxman, A.W., Emmett, B., Moldan, F., Gundersen, P., Schleppi, P., and Wright, R.F. 1998 Input-output budgets at the NITREX sites. *Forest Ecology and Management*, 101: 57-64.
Brown, D.J.A. 1983 Effect of calcium and aluminum concentrations on the survival of brown trout (Salmo trutta) at low pH. *Bulletin of Environmental Contamination and Toxicology*, 30(1): 582-587.
Brown, D.J.A., and Sadler, K. (1989) Fish Survival in Acid Waters. In Acid Toxicity and Aquatic Animals (P. Morris, A.J. Taylor, D.J.A. Brown, and J.A. Brown, eds), Cambridge: Cambridge University Press.
Buckton, Brewin, Lewis, Stevens, and Ormerod. 1998 The distribution of dippers, Cinclus cinclus (L.), in the acid sensitive region of Wales 1984-95. *Freshwater Biology*, 39(2): 387-396.
Clayton, J.L., and Menendez, R. 1996 Macroinvertebrate Responses to Mitigative Liming of Dogway Fork, West Virginia. *Restoration Ecology*, 4(3): 234-246.
Clenaghan, C., Giller, P.S., O'Halloran, J., and Hernan, R. 1998 Stream macroinvertebrate communities in a conifer-afforested catchment in Ireland: relationships to physico-chemical and biotic factors. *Freshwater Biology* 40(1): 175-193.
Collier, K.J., Ball, O.J., Graesser, A.K., Main, M.R., and Winterbourn, M.J. 1990 Do organic and anthropogenic acidity have similar effects on aquatic fauna? *Oikos*, 59(1): 33-38.
Cowan, D. (1998) Mitigation of headwater acidification in forests by use of planted riparian zones: a study in Northern England. . In Headwaters: Water Resources and soil conservation (M.J. Haigh, J. Krecek, G.S. Rajwar, and M.P. Kilmartin, eds), Rotterdam: AA Balkema.
Crossley, A., Wilson, D.B., and Milne, R. 1992 Pollution in the upland environment. *Environmental Pollution*, 75(1): 81-87.
Cruikshanks, R., Lauridsen, R., Harrison, A., Hartl, M.G.J., Kelly-Quinn, M., Giller, P.S., and O'Halloran, J. (2006) Evaluation of the use of the Sodium Dominance Index as a potential measure of acid sensitivity WATERAC Final Report, Dublin: Environmental Protection Agency and Council for Forest Research, pp 107.

Cruikshanks, R., Tierney, D., McGuinness, C., and Kelly-Quinn, M. in press Observations of recovery of the benthic macroinvertebrate community in an acid impacted upland stream in Ireland. . *XXIX Congress Lahti Finland 8 - 14 August 2004*, 29(3).
Cummins, T., Beier, C., Blanck, K., Visser, P.H.B., Farrell, E.P., Rasmussen, L., Kreutzer, K., Weis, W., Bredemeier, M., and Lamersdorf, N. 1995 The exman project—Biogeochemical fluxes in plantation forests on acid soils. *Water, Air, & Soil Pollution*, V85(3): 1653-1658.
Cummins, T., and Farrell, E.P. 2003 Biogeochemical impacts of clearfelling and reforestation on blanket-peatland streams: II. major ions and dissolved organic carbon. *Forest Ecology and Management*, 180(1-3): 557-570.
Dangles, O., Gessner, M.O., Guerold, F., and Chauvet, E. 2004a Impacts of stream acidification on litter breakdown: implications for assessing ecosystem functioning. *Journal of Applied Ecology*, 41(2): 365-378.
Dangles, O., Malmqvist, B., and Laudon, H. 2004b Naturally acid freshwater ecosystems are diverse and functional: evidence from boreal streams. *Oikos*, 104(1): 149-155.
Dangles, O.J., and Guérold, F.A. 2000 Structural and functional responses of benthic macroinvertebrates to acid precipitation in two forested headwater streams (Vosges Mountains, northeastern France). *Hydrobiologia*, V418(1): 25-31.
de Vries, W., Posch, M., Reinds, G.J., and Kämäri, J. (1993) Critical Loads and their Exceedance on Forest Soils in Europe. Rep. No. 58, Wageningen: DLO Winand Staring Centre.
Degerman, E., and Appelberg, M. 1992 The response of stream-dwelling fish to liming. *Environmental Pollution*, 78(1-3): 149-155.
Dollard, G.J., Unsworth, M.H., and Harve, M.J. 1983 Pollutant transfer in upland regions by occult precipitation. *Nature*, 302(5905): 241-243.
Donnelly, A., Jennings, E., and Allott, N. 2003 A review of liming options for afforested catchments in Ireland. *Biology and Environment: Proceedings of the Royal Irish Academy*, 103 (2): 91-105.
Driscoll, C.T., Baker, J.P., Bisogni, J.J., and Schofield, C.L. 1980 Effect of aluminium speciation on fish in dilute acidified waters. *Nature*, 284(5752): 161-164.
Erisman, J.W., Draaijers, G., Duyzer, J., Hofschreuder, P., Leeuwen, N.V., Romer, F., Ruijgrok, W., Wyers, P., and Gallagher, M. 1997 Particle deposition to forests--Summary of results and application. *Atmospheric Environment*, 31(3): 321-332.
Evans, C.D. 2005 Modelling the effects of climate change on an acidic upland stream. *Biogeochemistry*, 74: 21-46.
Evans, C.D., and Monteith, D.T. 2002 Natural and Anthropogenic Changes in The Chemistry of Six UK Mountain Lakes, 1988 to 2000. *Water, Air, & Soil Pollution: Focus*, V2(2): 33-46.
Evans, C.D., Monteith, D.T., and Harriman, R. 2001 Long-term variability in the deposition of marine ions at west coast sites in the UK Acid Waters Monitoring Network: impacts on surface water chemistry and significance for trend determination. *The Science of The Total Environment*, 265(1-3): 115-129.
Farrell, E.P. 1995 Atmospheric deposition in maritime environments and its impact on terrestrial ecosystems. *Water, Air, & Soil Pollution*, V85(1): 123-130.

Farrell, E.P., Aherne, J., Boyle, G.M., and Nunan, N. 2001 Long-Term Monitoring of Atmospheric Deposition and the Implications of Ionic Inputs for the Sustainability of a Coniferous Forest Ecosystem. *Water, Air, & Soil Pollution*, V130(1): 1055-1060.
Farrell, E.P., Cummins, T., and Boyle, G.M. (1997) AQUAFOR Report No. 1 Chemistry of Precipitation Throughfall and Soil Water, Cork, Wicklow and Galway Regions. In A study of the effects of stream hydrology and water quality in forested catchments on fish and invertebrates, Dublin: Coford.
Farrell, E.P., Van Den Beuken, R., Boyle, G.M., Cummins, T., and Aherne, J. 1998 Interception of seasalt by coniferous and broadleaved woodland in a maritime environment in Western Ireland. *Chemosphere*, 36: 985-987.
Ferrier, R.C., Helliwell, R.C., Cosby, B.J., Jenkins, A., and Wright, R.F. 2001 Recovery from acidification of lochs in Galloway, south-west Scotland, UK: 1979-1998. *Hydrology and Earth System Sciences* , 5(3): 421-432.
Fjellheim, A., and Raddum, G.G. 1990 Acid Precipitation - Biological Monitoring of Streams and Lakes. *Science of the Total Environment*, 96(1-2): 57-66.
Fjellheim, A., and Raddum, G.G. 1995 Benthic animal response after liming of three south Norwegian rivers. *Water, Air, & Soil Pollution*, V85(2): 931-936.
Forest-Service. (2000) Code of Best Practice (K.D. Collins, G. Gallagher, J. Gardiner, E. Hendrick, D. McAree, and D.G. Gallagher, eds), Dublin, Ireland: Forest Service, Department of the Marine and Natural Resources, pp 200.
Forest-Service. (2004) Afforestation Grant and Premium Schemes Manual, pp 135.
Forestry-Commission-UK. (2003) Forestry and Water Guidelines, Edinburgh: Forestry Commission, pp 66.
Fowler, D., Cape, J.N., Unsworth, M.H., Mayer, H., Crowther, J.M., Jarvis, P.G., Gardiner, B., and Shuttleworth, W.J. 1989 Deposition of Atmospheric Pollutants on Forests [and Discussion]. *Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences*, 324(1223): 247-265.
Friberg, N., and Jacobsen, D. 1999 Variation in growth of the detritivore-shredder Sericostoma personatum (Trichoptera). *Freshwater Biology*, 42(4): 625-635.
Friberg, N., Rebsdorf, A., and Larsen, S.E. 1998 Effects of Afforestation on Acidity and Invertebrates in Danish Streams and Implications for Freshwater Communities in Denmark. *Water, Air, & Soil Pollution*, V101(1): 235-256.
Gee, A.S., and Stoner, J.H. (1988) The effects of afforestation and acid deposition on the water quality and ecology of upland Wales. In Ecological Change in the Uplands, Oxford: Blackwell, pp 273-287.
Giller, P., O'Halloran, J., Kiely, G., Evans, J., Clenaghan, C., Hernan, R., Roche, N., and Morris, P. (1997) An Evaluation of the Effects of Forestry on Surface Water Quality and Ecology in Munster. In A study of the effects of stream hydrology and water quality in forested catchments on fish and invertebrates AQUAFOR Report 2), Dublin, Ireland: COFORD, pp 77.
Giller, P.S., O'Halloran, J., Hernan, R., Roche, N., Clenaghan, C., Evans, J., Kiely, G.K., Morris, P., Allott, N., Brennan, M., Reynolds, J., Cooke, D., Kelly-Quinn, M., Bracken, J., Coyle, S., and Farrell, E.P. 1993 An integrated study of forested catchments in Ireland. *Irish Forestry*, 50(1): 70-83.
Giller, P.S., and O'Halloran, J. 2004 Forestry and the aquatic environment: studies in an Irish context. *Hydrology and Earth System Sciences*, 8(3): 314-326.
Grace, J., and Unsworth, M. (1988) Climate and microclimate of the uplands. In Ecological change in the uplands. (M.B. Usher, and D.B.A. Thompson, eds), Edinburgh, UK: Blackwell Scientific Publications, pp 137–150.

Grieve, I.C., and Marsden, R.L. 2001 Effects of forest cover and topographic factors on TOC and associated metals at various scales in western Scotland. *Science of the Total Environment*, 265(1-3): 143-151.
Guerold, F., Boudot, J.-P., Jacquemin, G., Vein, D., Merlet, D., and Rouiller, J. 2000 Macroinvertebrate community loss as a result of headwater stream acidification in the Vosges Mountains (N-E France). *Biodiversity and Conservation*, V9(6): 767-783.
Hall, R.J. 1994 Responses of benthic communities to episodic acid disturbances in a lake outflow stream at the experimental lakes area, Ontario *Canadian Journal of Fisheries and Aquatic Sciences*, 51(8): 1877-1892.
Harriman, R., Anderson, H., and Miller, J.D. 1995 The role of sea-salts in enhancing and mitigating surface water acidity. *Water, Air, & Soil Pollution*, V85(2): 553-558.
Harriman, R., and Morrison, B.R.S. 1982 Ecology of streams draining forested and non-forested catchments in an area of central Scotland subject to acid precipitation. *Hydrobiologia*, V88(3): 251-263.
Harriman, R., Watt, A.W., Christie, A.E.G., Moore, D.W., McCartney, A.G., and Taylor, E.M. 2003 Quantifying the effects of forestry practices on the recovery of upland streams and lochs from acidification. *The Science of The Total Environment*, 310(1-3): 101-111.
Havas, M., and Rosseland, B.O. 1995 Response of zooplankton, benthos, and fish to acidification: An overview. *Water, Air, & Soil Pollution*, V85(1): 51-62.
Heath, R.H., Kahl, J.S., Norton, S.A., and Fernandez, J.J. 1992 Episodic acidification caused by the seasalt effect in coastal Maine streams. *Water Resources Research*, 28: 1081-1088.
Helliwell, R.C., Ferrier, R.C., Johnston, L., Goodwin, J., and Doughty, R. 2001 Land use influences on acidification and recovery of freshwaters in Galloway, south-west Scotland. *Hydrology and Earth System Sciences*, 5(3): 451-458.
Hesthagen, T. 1989 Episodic Fish Kills in an Acidified Salmon River in Southwestern Norway. *Fisheries Management and Ecology*, 14(3): 10-17.
Hettelingh, J.-P., Downing, R.J., and de Smet, P.A.M. (1991) Mapping Critical Loads for Europe, , Bilthoven.: RIVM.
Hindar, A., Henriksen, A., Ø, K., and Tørseth, K. 1995 Extreme acidification in small catchments in southwestern Norway associated with a sea salt episode. *Water, Air, & Soil Pollution*, V85(2): 547-552.
Hindar, A., Henriksen, A., Torseth, K., and Semb, A. 1994 Acid water and fish death. *Nature*, 372(6504): 327-328.
Hindar, A., Wright, R.F., Nilsen, P., Larssen, T., and Hogberget, R. 2003 Effects on stream water chemistry and forest vitality after whole-catchment application of dolomite to a forest ecosystem in southern Norway. *Forest Ecology and Management*, 180(1-3): 509-525.
Hirst, H., Chaud, F., Delabie, C., Ingrid Juttner, and J., O.S. 2004 Assessing the short-term response of stream diatoms to acidity using inter-basin transplantations and chemical diffusing substrates. *Freshwater Biology*, 49: 1072-1088.
Hornung, M., Brown, S.J., and Ranson, A. (1990a) Amlioration of surface water acidity by catchment management. In Acid Waters in Wales (R.W. Edwards, J.H. Stoner, and A.P. Gee, eds), Dordecht: Kluwer, Academic.
Hornung, M., Bull, K.R., Cresser, M., Ullyett, J., Hall, J.R., Langan, S., Loveland, P.J., and Wilson, M.J. 1995 The sensitivity of surface waters of Great Britain

to acidification predicted from catchment characteristics. *Environmental Pollution*, 87(2): 207-214.
Hornung, M., Le-Grice, S., Brown, N., and Hughes, S. (1990b) The role of geology and soils in controlling surface water acidity in Wales. In Acid Waters in Wales (R.W. Edwards, J.H. Stoner, and A.S. Gee, eds), Dordrecht : Kluwer Academic Publishers, pp 55-66.
Hornung, M., Reynolds, B., Stevens, P.A., and Hughes, S. (1990c) Water qualtiy changes from input to stream. In Acid Waters in Wales (R.W. Edwards, J.H. Stoner, and A.S. Gee, eds), Dordrecht: Kluwer Academic Publishers, pp 223-240.
Hornung, M., Reynolds, B., Stevens, P.A., and Neal, C. 1987 Stream acidification resulting from afforestation in the U.K. : evaluation of causes and possible ameliorative meaures. *Forest Hydrology and Watershed Management*, IAHS-AISH pubication No. 167.
Hornung, M., and Skeffington, R.A. (1993) Critical loads: concepts and applications. In ITE Symposium, London: HMSO.
Howells, G., Dalziel, T.R.K., Reader, J.P., and Solbe, J.F. 1990 EIFAC water quality criteria for European freshwater fish: Report on aluminium. *Chemistry and Ecology.*, 4(3): 117-173.
Hughes, S., Norris, D.A., Stevens, P.A., Reynolds, B., Williams, T.G., and Woods, C. 1994 Effects of forest age on surface drainage water and soil solution aluminium chemistry in stagnopodzols in Wales. *Water, Air, & Soil Pollution*, V77(1): 115-139.
Hurrell, J.W. 1995 Decadal Trends in the North Atlantic Oscillation: Regional Temperatures and Precipitation. *Science*, 269(5224): 676-679.
Ikuta, K., Munakata, A., Aida, K., Amano, M., and Kitamura, S. 2001 Effects of Low pH on Upstream Migratory Behavior in Land-Locked Sockeye Salmon Oncorhynchus nerka. *Water, Air, & Soil Pollution*, 130(1): 99-106.
Ikuta, K., Suzuki, Y., and Kitamura, S. 2003 Effects of low pH on the reproductive behavior of salmonid fishes. *Fish Physiology and Biochemistry*, V28(1): 407-410.
Jenkins, A., Cosby, B.J., Ferrier, R., Walker, T.A.B., and Miller, J.D. 1990 Modelling stream acidification in afforested catchments. An assessment of the relative effects of acid deposition and afforestation. *Journal of Hydrology*, 120: 163-181.
Jennings, E., Donnelly, A., and Allott, N. (2002) Effectiveness of buffer strips for the mitigation of acid runoff from afforested catchments. In Environmental RTDI Programme 2000 – 2006. , Wexford: Environmental Protection Agency, pp 28.
Kaste, Ø., and Skjelkvale, B.L. 2002 Nitrogen dynamics in runoff from two small heathland catchments representing opposite extremes with respect to climate and N deposition in Norway *Hydrology and Earth System Sciences*, 6(3): 351-362.
Keener, A.L., and Sharpe, W.E. 2005 The Effects of Doubling Limestone Sand Applications in Two Acidic Southwestern Pennsylvania Streams. *Restoration Ecology*, 13(1): 108-119.
Kelly-Quinn, M., Aherne, J., Ryan, D., and Farrell, E.P. (2000) Critical loads lake survey. Hydrochemical characteristics: 200 lake survey to determine the sensitivity of Irish lakes to acidification Forest Ecosystem Research Group, pp 25.

Kelly-Quinn, M., Tierney, D., Coyle, C., and Bracken, J.J. 1996a Factors affecting the susceptibility of Irish soft-water streams to forest-mediated acidification. *Fisheries Management and Ecology*, 3(4): 287-301.
Kelly-Quinn, M., Tierney, D., Roche, D., and Bracken, J.J. 1996b Distribution and abundance of trout populations in moorland and afforested upland nursery streams in County Wicklow. *Biology and Environment: Proceedings of the Royal Irish Academy*, 96B (3): 127-139.
Khanna, P.H., and Ulrich, B. (1984) Soil Characteristics Influencing Nutrient Supply in Forest Soils. In Nutrition of Plantation Forests (B.D. Bowen, and E.K.S. Nambiar, eds), London: Academic Press pp 79-118.
Kowalik, R.A., and Ormerod, S.J. 2006 Intensive sampling and transplantation experiments reveal continued effects of episodic acidification on sensitive stream invertebrates. *Freshwater Biology*, 51: 180-191.
Kratz, K.W., Cooper, S.D., and Melack, J.M. 1994 Effects of single and repeated experimental acid pulses on invertebrates in a high altitude Sierra Nevada stream. *Freshwater Biology*, 32(1): 161-183.
Kreutzer, K., Beier, C., Bredemeier, M., Blanck, K., Cummins, T., Farrell, E.P., Lammersdorf, N., Rasmussen, L., Rothe, A., de Visser, P.H.B., Weis, W., Wei, T., and Xu, Y.J. 1998 Atmospheric deposition and soil acidification in five coniferous forest ecosystems: a comparison of the control plots of the EXMAN sites. *Forest Ecology and Management*, 101(1-3): 125-142.
Kullberg, A. 1992 Benthic macroinvertebrate community structure in 20 streams of varying pH and humic content. *Environmental Pollution*, 78(1-3): 103-106.
Langan, S. 1987 Episodic acidification of streams at Loch Dee, SW Scotland *Transactions of the Royal Society of Edinburgh. Earth sciences*, 78(4): 393-397.
Langan, S.J., and Hirst, D. 2004 An analysis of the long-term variation in stream water quality for three upland catchments at Loch Dee (Galloway, S.W. Scotland) under contrasting land management. *Hydrology and Earth System Sciences*, 8(3): 422-435.
Larssen, T., and Holme, J. 2006 Afforestation, seasalt episodes and acidification - A paired catchment study in western Norway. *Environmental Pollution*, 139(3): 440-450.
Ledger, M.E., and Hildrew, A.G. 2000 Herbivory in an acid stream. *Freshwater Biology*, 43(4): 545-556.
Lees, F.M. 1995 Have emission reductions led to improvements in water quality at Loch Dee in Galloway? *Water, Air, & Soil Pollution*, V85(2): 695-700.
LeFevre, S.R., and Sharpe, W.E. 2002 Acid Stream Water Remediation Using Limestone Sand on Bear Run in Southwestern Pennsylvania. *Restoration Ecology*, 10(2): 223-236.
Lehane, B.M., Giller, P.S., O'Halloran, J., and Walsh, P.M. 2004 Relative influences of catchment geology, land use and in-stream habitat on brown trout populations in South-western Ireland. *Biology and Environment: Proceedings of the Royal Irish Academy*, 104B(1): 43-54.
Lepori, F., Barbieri, A., and Ormerod, S.J. 2003 Effects of episodic acidification on macroinvertebrate assemblages in Swiss Alpine streams. *Freshwater Biology*, 48(10): 1873-1885.
Lepori, F., and Ormerod, S.J. 2005 Effects of spring acid episodes on macroinvertebrates revealed by population data and in situ toxicity tests. *Freshwater Biology*, 50(9): 1568-1577.

Lindberg, S.E., and Owens, J.G. 1992 Throughfall studies of deposition to forest edges and gaps in montane ecosystems. *Biogeochemistry*, V19(3): 173-194.
Lingdell, P.E., and Engblom, E. 1995 Liming restores the benthic invertebrate community to "pristine" state. *Water, Air, & Soil Pollution*, V85(2): 955-960.
McKie, B.G., Petrin, Z., and Malmqvist, B. 2006 Mitigation or disturbance? Effects of liming on macroinvertebrate assemblage structure and leaf-litter decomposition in the humic streams of northern Sweden. *Journal of Applied Ecology*, 43: 780-791.
Melody, K.J., and John, S.R. 2004 Responses of invertebrates and algae of a boreal coniferous forest stream to experimental manipulation of leaf litter inputs and shading. *Hydrobiologia*, V519(1): 197-206.
Milner, N.J. (1989) Strategies to minimise the impact of forestry on fisheries. In Proceedings of the Institute of Fisheries Management 20th Annual Study Course (T. Cross, and B. Ottoway, eds), Regional Technical College Galway, pp 212-225.
Neal, C., Ormerod, S.J., Langan, S.J., Nisbet, T.R., and Roberts, J. 2004a Sustainability of UK Forestry : contemporary issues for the protection of freshwaters, a conclustion. *Hydrology and Earth System Sciences*, 8(3): 589-595.
Neal, C., and Reynolds, B. (1998) The impact of conifer harvesting and replanting on upland water quality, Bristol: Envionment Agency Research and Development, pp 137.
Neal, C., Reynolds, B., Adamson, J.K., Stevens, P.A., Neal, M., Harrow, M., and Hill, S. 1998 Analysis of the impacts of major onion variations on surface water acidity particularly with regard to conifer harvesting: case studies from Wales and Northern England. *Hydrology and Earth System Sciences*, 2(2/3): 303-322.
Neal, C., Reynolds, B., Neal, M., Pugh, B., Hill, L., and Wickham, H. 2001 Long-term changes in the water quality of rainfall, cloud water and stream water for moorland, forested and clear-felled catchments at Plynlimon, mid-Wales. *Hydrology and Earth System Sciences*, 5(3): 459-476.
Neal, C., Reynolds, B., Neal, M., Wickham, H., Hill, L., and Pugh, B. 2003 The Impact of Conifer Harvesting on Stream Water Quality: A Case Study in Mid-Wales. *Water, Air, & Soil Pollution: Focus*, V3(1): 119-138.
Neal, C., Reynolds, B., Neal, M., Wickham, H., Hill, L., and Williams, B. 2004b The impact of conifer harvesting on stream water quality: the Afon Hafren, mid-Wales. *Hydrology and Earth System Sciences*, 8(3): 503-520.
NEGTAP. (2001) National Expert Group on Transboundary Air Pollution: Acidification, Eutrophication and Ground Level Ozone in the UK. First Report, Retrieved from: www.nbu.ac.uk/negtap/finalreport.htm.: UK Department of the Environment, Transport and the regions and the devolved administrations.
Nilsson, J., and Grennfelt, P. (1988) Critical loads for sulphur and nitrogen. , Copenhagen, Denmark UNECE/Nordic Council of Ministers, pp 418.
Nilsson, S.I., Miller, H.G., and Miller, J.D. 1982 Forest Growth as a Possible Cause of Soil and Water Acidification: An Examination of the Concepts. *Oikos*, 39(1): 40-49.
Nisbet, T.R. (2001) Forestry Operations - A review of the best practice guidance for the protection of the freshwater environment. R&D Technical Report P210, Swindon, Wilts.: Environment Agency.

Nisbet, T.R., Fowler, D., and Smith, R.I. 1995 An investigation of the impact of afforestation on stream-water chemistry in the Loch Dee catchment, SW Scotland. *Environmental Pollution*, 90(1): 111-120.
Niyogi, D.K., McKnight, D.M., and Lewis, W.M., Jr. 1999 Influences of Water and Substrate Quality for Periphyton in a Montane Stream Affected by Acid Mine Drainage. *Limnology and Oceanography*, 44(3): 804-809.
Oliver, B.G., Thurman, E.M., and Malcolm, R.L. 1983 The contribution of humic substances to the acidity of colored natural waters. *Geochimica et Cosmochimica Acta*, 47(11): 2031-2035.
Ormerod, S.J., Allenson, N., Hudson, D., and Tyler, S.J. 1986 The distribution of breeding dippers (Cinclus cinclus (L.); Aves) in relation to stream acidity in upland Wales. *Freshwater Biology*, 16(4): 501-507.
Ormerod, S.J., Donald, A.P., and Brown, S.J. 1989 The influence of plantation forestry on the pH and aluminium concentration of upland welsh streams: A re-examination. *Environmental Pollution*, 62(1): 47-62.
Ormerod, S.J., Jones, M.E., Jones, M.C., and Phillips, D.R. 2004 The effects of riparian forestry on invertebrate drift and brown trout in upland streams of contrasting acidity. *Hydrology and Earth System Sciences*, 8(3): 578-588.
Ormerod, S.J., O'Halloran, J., Gribbin, S.D., and Tyler, S.J. 1991a The Ecology of Dippers Cinclus cinclus in Relation to Stream Acidity in Upland Wales: Breeding Performance, Calcium Physiology and Nestling Growth. *The Journal of Applied Ecology*, 28(2): 419-433.
Ormerod, S.J., Rundle, S.D., Lloyd, E.C., and Douglas, A.A. 1993 The influence of riparian management on the habitat structure and macroinvertebrate communities of upland streams draining plantation forests. *Journal of Applied Ecology*, 30(1): 13-24.
Ormerod, S.J., Rutt, G., Weatherley, N.S., and Wade, K. (1991b) Detecting and managing the influence of forestry on river systems in Wales : Results from surveys, experiments and models In Irish Rivers: Biology and Management (M.W. Steer, ed, Dublin: Royal Irish Academy, pp 163-184.
Ormerod, S.J., and Wade, K.R. (1990) The role of acidity in the ecology of Welsh lakes and streams. In Acid Waters in Wales (R.W. Edwards, J.H. Stoner, and A.S. Gee, eds), Dordrecht: Kluwer Academic Publishers, pp 95-109.
Ormerod, S.J., Wade, K.R., and Gee, A.S. 1987 Macrofloral assemblages in upland Welsh streams in relation to acidity, and their importance to invertebrates. *Freshwater Biology*, 18: 545–557.
Posch, M., de Smet, P.A.M., Hettelingh, J.-P., Downing, R.J., and Editors. (1995) Calculation and mapping of critical thresholds in europe, status report, Bilthoven: Co-ordination Centre for Effects, RIVM
Puhr, C.B., Donoghue, D.N.M., Stephen, A.B., Tervet, D.J., and Sinclair, C. 2000 Regional Patterns of Streamwater Acidity and Catchment Afforestation in Galloway, SW Scotland. *Water, Air, & Soil Pollution*, V120(1): 47-70.
Raddum, G.G., and Fjellheim, A. 2003 Liming of River Audna, Southern Norway: A Large-scale Experiment of Benthic Invertebrate Recovery. *AMBIO: A Journal of the Human Environment*, 32(3): 230-234.
Ranger, L.A.a.J. 2001 Impact of tree species on soil solutions in acidic conditions. *Annals of Forest Science*, 58 47-58.
Reader, J.P., and Dempsey, C.H. (1989) Episodic changes in water chemistry and their effects on fish. In Acid toxicity and aquatic animals (P. Morris, A.J.

Taylor, D.J.A. Brown, and J.A. Brown, eds), Cambridge: Cambridge Publications.
Rees, R.M., and Ribbens, J.C.H. 1995 Relationships between afforestation, water chemistry and fish stocks in an upland catchment in south west Scotland. *Water, Air, & Soil Pollution*, V85(2): 303-308.
Reynolds, B., Neal, C., and Norris, D.A. 2001 Evaluation of regional acid sensitivity predictions using field data: issues of scale and heterogeneity. *Hydrology and Earth System Sciences*, 5(1): 75-81.
Reynolds, B., Ormerod, S.J., and Gee, A.S. 1994 Spatial patterns concentrations in upland Wales in relation to catchment forest cover and forest age. *Environmental Pollution*, 84(1): 27-33.
Reynolds, B., and Stevens, P.A. 1998 Assessing soil calcium depletion following growth and harvesting of Sitka spruce plantation forestry in the acid sensitive Welsh uplands *Hydrology and Earth System Sciences*, 2(2/3): 345-352.
Reynolds, B., Wood, M.J., Truscott, A.M., Brittain, S.A., and Williams, D.L. 2000 Cycling of nutrient base cations in a twelve year old Sitka spruce plantation in upland mid-Wales. *Hydrology and Earth System Sciences*, 4(2): 311 - 321.
Rosemond, A.D., Reice, S.R., Elwood, J.W., and Mulholland, P.J. 1992 The effects of stream acidity on benthic invertebrate communities in the south-eastern United States. *Freshwater Biology*, 27(2): 193-209.
Rosseland, B.O. 1986 Ecological effects of acidification on tertiary consumers. Fish population responses. *Water, Air, & Soil Pollution*, 30(1): 451-460.
Rosseland, B.O., and Skogheim, O.K. 1987 Differences in sensitivity to acidic soft-water among strains of brown trout (Salmo trutta L). . *Annales De La Societe Royale Zoologique De Belgique*, 117(S1): 255-264.
Sayer, M.D.J., Reader, J.P., and Dalziel, T.R.K. 1993 Freshwater acidification: effects on the early life stages of fish. *Reviews in Fish Biology and Fisheries*, 3(2): 95-132.
Schofield, C.L. 1976 Acid Precipitation: Effects on fish. *Ambio*, 5(5-6): 228-230.
Simmons, K.R., and Doyle, K. 1996 Limestone Treatment of Whetstone Brook, Massachusetts. III. Changes in the Invertebrate Fauna. *Restoration Ecology*, 4(3): 284-292.
Smiddy, P., O'Halloran, J., O'Mahony, B., and Taylor, A.J. 1995 The breeding biology of the dipper Cinclus cinclus in South-west Ireland *Bird Study*, 42(1): 76-81.
Stevens, P.A., Reynolds, B., Hughes, S., Norris, D.A., and Dickinson, A.L. 1997 Relationships between spruce plantation age, solute and soil chemistry in Hafren forest. *Hydrology and Earth System Sciences*, 1: 627-637.
Stoner, J., Gee, A., and Wade, K. 1984 The effects of acidification on the ecology of streams in the upper Twyi catchment in West Wales. *Environmental Pollution, A*, 36: 125-157.
Sutcliffe, D.W., and Hildrew, A.G. (1989) Invertebrate communties in acid streams. In Acid Toxicity and aquatic animals (R. Morris, E.W. Taylor, D.J.A. Brown, and J.A. Brown, eds), Cambridge: Cambridge University Press, pp 13-29.
Teien, H.-C., Salbu, B., Kroglund, F., and Rosseland, B.O. 2004a Transformation of positively charged aluminium-species in unstable mixing zones following liming. *Science of the Total Environment*, 330(1-3): 217-232.
Teien, H.-C., Standring, W.J.F., Salbu, B., Marskar, M., and Hindar, F.K.a.A. 2004b Mobilization of aluminium and deposition on fish gills during sea salt

episodes – catchment liming as countermeasure. *Journal of Environmental Monitoring*, 6: 191-200.
Thiébaut, G., and Muller, S. 1999 A macrophyte communities sequence as an indicator of eutrophication and acidification levels in weakly mineralised streams in north-eastern France. *Hydrobiologia*, V410(0): 17-24.
Thomsen, A.G., and Friberg, N. 2002 Growth and emergence of the stonefly Leuctra nigra in coniferous forest streams with contrasting pH. *Freshwater Biology*, 47: 1159-1172.
Tierney, D., Kelly-Quinn, M., and Bracken, J.J. 1998 The faunal communities of upland streams in the eastern region of Ireland with reference to afforestation impacts. *Hydrobiologia*, V389(1): 115-130.
Toner, P., Bowman, J., Clabby, K., Lucey, J., McGarrigle, M., Concannon, C., Clenaghan, C., Cunningham, P., Delaney, J., O'Boyle, S., MacCárthaigh, M., Craig, M., and Quinn, R. (2005) Water Quality in Ireland Report 2001-2003, Johnstown Castle, Co. Wexford, Ireland: Environmental Protection Agency, pp 232.
Turnpenny, A.W.H., Sadler, K., Aston, R.J., Milner, A.G.P., and Lynam, S. 1987 The fish populations of some streams in Wales and northern England in relation to acidity and associated factors. *Journal of Fish Biology*, 31(3): 415-434.
Unsworth, M.H. 1984 Evaporation from forests in cloud enhances the effects of acid deposition. *Nature*, 312(5991): 262-264.
van Breemen, N., Mulder, J., and Driscoll, C.T. 1983 Acidification and alkalinization of soils. *Plant and Soil*, V75(3): 283-308.
Ventura, M., and Harper, D. 1996 The impacts of acid precipitation mediated by geology and forestry upon upland stream invertebrate communities. *Archiv für Hydrobiologie*, 138(2): 161-173.
Wade, K.R., Ormerod, S.J., and Gee, A.S. 1989 Classification and ordination of macroinvertebrate assemblages to predict stream acidity in upland Wales. *Hydrobiologia*, V171(1): 59-78.
Wang, X., Burns, D.A., Yanai, R.D., Briggs, R.D., and Germain, R.H. 2006 Changes in stream chemistry and nutrient export following a partial harvest in the Catskill Mountains, New York, USA. *Forest Ecology and Management*, 223(1-3): 103-112.
Waters, D., and Jenkins, A. 1992 Impacts of afforestation on water quality trends in two catchments in mid-Wales. *Environmental Pollution*, 77(2-3): 167-172.
Weatherley, N.S. 1988 Liming to mitigate acidification in freshwater ecosystems: A review of the biological consequences. *Water, Air, & Soil Pollution*, V39(3): 421-437.
Weathers, K.C., Cadenasso, M.L., and Pickett, S.T.A. 2001 Forest Edges as Nutrient and Pollutant Concentrators: Potential Synergisms between Fragmentation, Forest Canopies, and the Atmosphere. *Conservation Biology*, 15(6): 1506-1514.
Weston, D.W. 1995 The effectiveness of buffer zones in conifer-afforested catchments. *Journal of the Chartered Institution of Water and Environmental Management*, 9(4): 396-404.
Wigington, P.J., Jr., DeWalle, D.R., Murdoch, P.S., Kretser, W.A., Simonin, H.A., Sickle, J.V., and Baker, J.P. 1996 Episodic Acidification of Small Streams in the Northeastern United States: Ionic Controls of Episodes. *Ecological Applications*, 6(2): 389-407.

Wood, C. (1989) The physiological problems of fish in acid waters. In Acid Toxicity and Aquatic Animals (P. Morris, A.J. Taylor, D.J.A. Brown, and J.A. Brown, eds), Cambridge: Cambridge University Press.

Wright, J.F. 1995 Development and use of a system for predicting the macroinvertebrate fauna in flowing waters *Australian journal of ecology*, 20(1): 181-197.

Wright, R.F., Norton, S.A., Brakke, D.F., and Frogner, T. 1988 Experimental verification of episodic acidification of freshwaters by sea salts. *Nature*, 334(6181): 422-424.