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Western River Basin District

Programme of Measures

Forest and Water National Study

Forests and Surface water Eutrophication - Sedimentation Literature Review

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Literature Review

An evaluation of the role of forests and forest practices in the eutrophication and sedimentation of receiving waters

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1. INTRODUCTION

The Water Framework Directive (WFD) (2000/60/EC) establishes an integrated and co-ordinated framework for the sustainable management of water. Its purposes include ensuring “enhanced protection and improvement of the aquatic environment", promoting sustainable water use, and preventing further deterioration of water bodies. It also requires that rivers, lakes, estuaries, coastal waters and groundwater achieve and /or maintain at least ‘good status’ by 2015 (European Union, 2005a).

In implementing the WFD, the process of ‘risk assessment’ entails identification of environmental conditions, design of monitoring programmes, and the formulation of appropriate, cost effective protection and improvement measures (European Union, 2003). This methodology follows the principles outlined in a United Kingdom Technical Advisory Drafting Group paper (see UKTAG, 2003). The concept of ‘pressure-pathway-receptor’ is used as the framework in risk assessment and requires an understanding of the relationships between a pressure (such as physical or chemical habitat degradation) and the mechanism (or pathway) by which the pressure exerts its impact on an ecosystem (the receptor) (European Union, 2003). Pressures on freshwaters in Western Europe include 1) direct inputs of organic and inorganic pollutants from point sources 2) physical habitat alteration and 3) diffuse input of pollutants from surrounding catchments.

Forestry in Ireland, amongst other landuses, is recognised as a potential source of diffuse pollution, in conjunction with many others, to water courses and represents a risk to the ecological integrity of running waters (Duggan *et al.*, 2000; Gallagher *et al.*, 2000; Johnson *et al.*, 2000; Giller *et al.*, 1997). Forests currently cover approximately 10.15% of the Irish landscape, of which 77% is predominantly coniferous, 13% is broadleaved, 4% mixed forest and 6% other wooded areas (Forestry & Timber Yearbook, 2006). There has been a shift away from conifer planting in recent years with approximately 30% of new forest planting in 2005 being of broadleaved forest (Forestry & Timber Yearbook, 2007). Although the land now being planted/afforested is generally of better quality and located at lower altitudes, the majority of Irish forests have historically been planted on agriculturally unproductive land. Much of this land is in the uplands, such that many Irish rivers either rise in or receive drainage from these upland catchments. While recognising other landuse/catchment pressures, this review focuses on the concern of forests and forest operations and the pressures of eutrophication and sedimentation.

2. PRESSURE - EUTROPHICATION

Eutrophication is the process whereby a body of water becomes over-enriched with nutrients, in particular nitrogen (N) and phosphorus (P). This over-enrichment results in accelerated growth of algae and other plant life which in turn can deplete oxygen levels in the water, leading to the loss of aquatic animals.

Anthropogenic nutrient inputs to rivers and lakes can be through point or non-point (diffuse) sources. At a point source, such as a sewage treatment plant, the measurement of discharge and chemical concentrations can be monitored at a single location. Consequently, point sources can often be controlled by treatment at the source. Non-point sources of pollution may be more temporally variable and linked to seasonal activity (e.g. forest operations) or episodic events, such as heavy rainfall (Carpenter *et al.*, 1998). Non-point inputs may also reach receiving waters via multiple routes making them difficult to measure and regulate and rely on land management practices to control pollutants.

Eutrophication caused by excessive inputs of N and P affects 38% of the surface waters in Ireland (less than 1% were subject to a serious degree of pollution). Agriculture is estimated to be the major source of these pollutants to Irish rivers and estuaries, with 70% of P loads and 82% of N loads, respectively, attributed to agricultural sources (Toner *et al.*, 2005). Despite their relatively small contribution to the overall N and P loading to running waters, commercial forests may have a disproportionately high eutrophication impact. They are commonly located in upland areas where their effluent waters drain into small headwater tributaries. Giller & Malmqvist, 1998). These headwaters may provide a source of high quality water for public and private supply, support important salmonid fisheries and are often of a high conservation value (Nisbet, 2002).

In contrast to other countries, Ireland applies very small amounts of N and potassium (K) fertiliser to forests (e.g. 11% and 2% respectively of the total fertiliser applied in 2006; pers. comm Coillte, 2007), with P being the major nutrient applied. Although K has been shown to leach readily from fertilised catchments (73% on first year after application in Scotland; Harriman, 1978), it is widely believed to play an insignificant role as a limiting nutrient for plant growth in natural waters (Webster & Patten, 1979) and because of its negligible use in Ireland will not be discussed further. As the transport mechanism and availability of P and N differ greatly, they will be dealt with separately below.

2.1 THE INDIVIDUAL ELEMENTS OF THE PRESSURE

2.1.1 Phosphorus

Phosphorus is often found to be the limiting nutrient in inland fresh waters (Carpenter *et al.*, 1998; Smith *et al.*, 1999; Wade *et al.*, 2001). 'Limiting' here means that P is the resource in shortest supply, relative to demand, so that increases or decreases in P have a direct effect on plant biomass. Values in excess of 30 µg/l ortho-phosphate phosphorus (PO_4–P) in river waters and in excess of 20 µg/l total P (P_{tot}) in lakes are considered by the Irish EPA to lead to eutrophication (Lucey *et al.*, 1999).

The Phosphorus Regulations, 1998, is a legislative measure aimed at reducing eutrophication in rivers and lakes in Ireland. The targets set by the Phosphorus Regulations are designed to prevent deterioration of waters of good quality and to improve waters of unsatisfactory quality to a specified standard (see Box 1).

Box 1 Standards for Phosphorus in 1) rivers and 2) lakes (from Toner *et al.* (2005))

1) **Rivers**

The annual median concentration of molybdate reactive phosphate (MRP) shall not exceed:

a) 15 µg P/l in Q5 waters (unpolluted)
b) 20 µg P/l in Q4-5 waters (unpolluted)
c) 30 µg P/l in Q4 waters (unpolluted)
d) 50 µg P/l in Q3-4 waters (slightly polluted)
e) 70 µg P/l in Q3 waters (moderately polluted)

or

f) that existing satisfactory biological quality conditions (i.e., Q5, Q4-5 and Q4) be maintained and
g) that less than satisfactory biological conditions (Q3-4 or less) be improved. In general the improvement required is of half a quality rating (e.g., Q3-4 to Q4) but seriously polluted waters (Q2 or less) must be restored to Q3 as a minimum requirement.

2) **Lakes**

The annual mean concentration of total phosphorus (TP) shall fall within the ranges:
a) <5 µg P/l in Ultra-Oligotrophic lakes
b) >5 to <10 µg P/l in Oligotrophic lakes
c) >10 to <20 µg P/l in Mesotrophic lakes
d) >20 to <50 µg P/l in Eutrophic lakes.

or

e) that existing satisfactory biological quality conditions (defined as Ultra-Oligotrophic, Oligotrophic and Mesotrophic status) be maintained and
f) that unsatisfactory biological conditions (Eutrophic, Hypertrophic) be improved as follows - Eutrophic waters to achieve Mesotrophic status and Hypertrophic waters to achieve Eutrophic status.

The most available form of P in streamwater is orthophosphate, which is transported to aquatic ecosystems as soluble and particulate P. Soluble P is readily available to plants and is usually termed dissolved (molybdate) reactive P (DRP) or soluble reactive P (SRP) (House & Denison, 1998). Particulate P (PP), bound to suspended soil particles and dissolved organic matter, forms a long-term reserve of low biological availability, mainly in lakes. (Sharpley *et al.*, 1994). The term 'total P' (TP) represents both DRP and PP. The P fraction often quoted in the literature is total reactive phosphorus (TRP), also called molybdate reactive phosphorus (MRP), which represents the amount of P potentially available for plants and consists, in unknown amounts, of free orthophosphate, phosphate displaced from acid labile inorganic colloids or particles, and phosphate released by the hydrolysis of acid labile phosphate esters (Åström *et al.*, 2002).

Under natural conditions, P is present in low quantities. The total amount of P found in unfertilised topsoils ranges from 50 – 1100 mg kg^{-1} (Morgan, 1997) while the rate of chemical weathering of P from rock is of the order of 0.01-5.0 kg P ha^{-1} $year^{-1}$ (Newman, 1995). Concentrations of soluble P in the soil are very low, mostly varying from 0.03 to 0.2 mg l^{-1}. This low concentration, combined with the small volume of soil solution, means that the amount of soluble soil P available to plants must be replenished many times in a growing season through the slow release of P bound in soils in organic or inorganic forms (Steén, 1997). Soil P concentrations usually decrease significantly with depth. For example, in the catchment of the River Trent, UK, the mean total P concentration at the soil surface was over 115 µg P g^{-1}, whereas that at 150cm depth was around 40 µg P g^{-1} (Dils & Heathwaite, 1996; cited in Heathwaite, 1997). Regardless of soil depth, the water-extractable fraction formed only a very small proportion (< 0.5%) of total soil P; hence leaching losses of soil P under natural conditions are low, but this depends on the extent to which the soil is saturated with regard to P. It should be noted that the publications referenced above relate to agricultural mineral soils which are not typical of conifer forest soil conditions in Ireland.

The ability of soils to retain P is an important factor in relation to potential stream eutrophication. The availability of soluble P in mineral soils is controlled mainly by chemical processes, principally by the naturally high concentrations of free iron and aluminium oxides and hydrous oxides in the soil (Sharpley, 1995; Morgan, 1997; Leinweber *et al.*, 1999; Daly *et al.*, 2000; Reynolds & Davies, 2001). In addition, amorphous aluminium hydroxy compounds may be located in interlayer sites of supple aluminium silicates (Morgan, 1997). These materials are highly efficient in adsorbing $H_2PO_4^-$ ions that may be present in the soil solution, and retention occurs as a result of exchange between the $H_2PO_4^-$ anion and hydroxyl (OH^-) ions associated with the Fe and/or Al. Under alkaline conditions in the presence of free calcium carbonate ($CaCO_3$), adsorption of $H_2PO_4^-$ /HPO_4^{2-} on to calcite can also occur by replacement of water, bicarbonate (HCO_3^-) or OH^- ions present on the calcite particles (Morgan, 1997). Parallel with these adsorption reactions, $H_2PO_4^-$ ions in solution may undergo precipitation reactions, the nature of which vary with the pH of the soil. Under acid conditions (pH < 5.0), the presence of reactive Al and Fe may result in the formation of poorly soluble hydroxy metal phosphates (e.g. $Al(OH)_2H_2PO_4$). In contrast, under alkaline conditions, presence of reactive calcium causes precipitation of dicalcium phosphate anhydrous (Morgan, 1997). The concentration of Fe and Al in solution in mineral soils is controlled by pH and they are most soluble under acid conditions (Heathwaite, 1997). Fe and Al oxides occur as discrete particles or as coatings on other soil particles, especially clay. Soil texture is therefore important because the finer the texture (i.e. the higher the clay fraction) the greater the surface area of the soil available for sorption. In general, acid mineral soils will tend to immobilize P, while peaty soils, with their very low concentrations of Fe and Al, will tend to leach P. As many forests in Ireland have been established on low productivity peaty soils (44%; see Table 1), these forests could potentially contribute a higher risk to the eutrophication of nearby watercourses. Phosphorus may, however, still be lost from soils with a high P-retention capacity if they are prone to surface run-off and erosion (Heathwaite, 1997).

Table 1. Percent of soil types on Coillte's estate (Anon, 1999)

Soil type	**%**
Gley	20
Podsol	20
Brown Earth	11
Other Mineral	5
Virgin Blanket Bog	34
Cutover Raised Bog	4
Cutover Blanket Bog	3
Virgin Raised Bog	2
Fen	1
Total	100

2.1.2 Nitrogen

Surface and groundwaters used for domestic water supply that contain high levels of nitrate represent a public health risk. A limit of 50 mg l^{-1} of nitrate (equivalent to 11.3 mg l^{-1} nitrate-nitrogen) in water sources has been implemented in EU directives and related national regulations in order to minimise this risk. In addition, a guide level of 25 mg l^{-1} of nitrate (or 5.65 mg l^{-1} nitrate-nitrogen) was outlined in the 1980 directive and is recommended as an indication of significant contamination (Toner *et al.*, 2005).

Currently no studies exist on whether N is a limiting factor in upland rivers in Ireland. N–driven eutrophication of benthic algae may potentially occur during low flow conditions in summer, when supply of nutrients is reduced and denitrification rates are highest (Maberly *et al.*, 2004). Since many upland lakes are limited or co–limited by nitrogen it is reasonable to assume that upland streams and river will be similarly limited. One uncertain issue is whether or not dissolved organic nitrogen is available as a nitrogen source for aquatic plants and algae in freshwaters. Preliminary evidence from lakes suggests that at least some forms of organic nitrogen are available to phytoplankton (unpublished data cited in Maberly *et al.*, 2004).

Nitrogen is an essential nutrient as it is a component of proteins and DNA. It is also, following carbon, hydrogen and oxygen, the resource required in greatest quantities by algae and plants (Hecky & Kilham, 1988). In water, nitrogen is potentially available in various forms as inorganic species such as nitrate (NO_3–N), nitrite (NO_2–N), ammonia (NH_3–N), ammonium (NH_4–N) and di–nitrogen (N_2), though this latter form is only available to particular nitrogen-fixing cyanobacteria (Reynolds, 1984; cited in Maberly *et al.*, 2004). Urea ($CO(NH_2)_2$) is a soluble organic compound containing nitrogen that is released in urine and applied to land as fertilizer and easily degrades into inorganic forms of nitrogen (Maberly *et al.*, 2004).

Natural ecosystems are characterised by a tight internal cycling of N. Leaching losses and gaseous losses are generally less than a few kg N ha^{-1} $year^{-1}$ (Gundersen & Bashkin, 1994). High leaching losses may, however, occur after a disturbance of the system.

Inputs of N by biological fixation are generally small in forest ecosystems compared to atmospheric deposition inputs, except for ecosystems with N fixing plant species (e.g. alder forests) (Boring *et al.*, 1988). Although alder is a significant component of broadleaf afforestation in Ireland, there are very few alder forests, with the species being generally found as single trees, rows along river banks, or small blocks within forests (pers. comm, Forest Service). As a result, inputs of N by biological fixation are most likely to be minimal in Irish forests. Almost all nitrogen in the soil is associated with organic matter. Including the humus horizon, the soil organic matter contains the largest pool of nitrogen in a boreal forest (Nihlgård *et al.*, 1994). A small proportion of the total is mineralized, principally to ammonium and nitrate. Nitrification is a crucial process for N losses by nitrate leaching and denitrification. Nitrate is relatively mobile in soils and if it is not quickly taken up by plants or microorganisms, it is easily leached by percolating water. Leaching of ammonium, on the other hand, rarely occurs. If it is not assimilated by plants or microorganisms, it may be nitrified, adsorbed on the exchange complex, immobilized in the lattice of clay minerals, or denitrified.

The status and fluxes of N in forest ecosystems are strongly regulated by rates of N mineralization and immobilization. The influences of these processes on biogeochemical cycles at a catchment scale have been demonstrated in experiments at Hubbard Brook Experimental Forest (HBEF), New Hampshire, USA. Over a three year period after clearcutting a hardwood forest at Hubbard Brook, forest-floor organic matter decreased by 10800 kg ha^{-1}, soil organic matter declined by 18900 kg ha^{-1} and net N loss from the soil was estimated to be 472 kg ha^{-1} with an increased export of inorganic N in the stream of 337 kg ha^{-1} (Likens *et al.*, 1970). However, the extent to which this experimental study was undertaken is not reflective of forest operations elsewhere. In a study on blanket peatland in western Ireland, Cummins & Farrell (2003) found increases in nitrate after felling and/or fertiliser treatments only in a few cases. While the maximum concentration recorded was 2.5 mg NO_3^- l^{-1} (0.6 mg N l^{-1}), in other study sites, following clearfelling of a considerable area of the catchment, including land upstream and directly surrounding the sampling point, no significant increase in NO_3^- concentrations were recorded (Cummins & Farrell, 2003). In a review of catchments in Britain, Neal *et al.* (1998) found that higher concentrations of NO_3^- occurred in a small number of sites post-felling, and several years after felling a decline in concentration was observed. The authors concluded that forest harvesting could have distinct local effects on certain chemical components of runoff for the first four years after felling, but was confined to a small number of sites where NO_3^- production and aluminium leaching were high. Studies in Plynlimon and Beddgelert in Wales showed increases in NO_3^- concentrations post-felling (Reynolds & Edwards, 1995). A maximum of 3.2 mg N l^{-1} was observed at Plynlimon approximately 1 year after the start of felling with concentrations declining to control stream values (0.5 mg N l^{-1}) after 5 years. A similar response was observed at Beddgelert, where NO_3^- concentrations in felled catchments were higher than those in the unfelled control catchment for 3 years, before declining to below control levels. Other studies have shown N losses from sites in North America and northern Scandinavia of < 1.4 kg ha^{-1} $year^{-1}$, while sites in southern Scandinavia and Central Europe have exhibited loss rates often >7 kg ha^{-1} $year^{-1}$ (Nihlgård *et al.*, 1994).

There has been increasing concern that increased anthropogenic inputs of N coupled with forest maturation may lead to the phenomenon of nitrogen saturation where N inputs exceed the demand of both the vegetation and the microbes, which results in nitrification and nitrate leaching (Aber *et al.*, 1989). Over an 8 year monitoring period of a first-rotation Sitka spruce (*Picea sitchensis* (Bong.) Carr.) plantation on a spodosol soil (derived from schist and quartzite bedrock) at Roundwood in eastern Ireland, Farrell *et al.* (2001) found that N output fluxes (deep soil water) exceeded N input fluxes (total deposition) suggesting that this area was N saturated in accordance with the definition of Aber *et al.* (1989). This was also evident in the major streams in the Plynlimon catchment in Wales where N output fluxes were an order of magnitude higher on both podzolic and gley soils. However this was only apparent for a small drainage ditch on gley soils, with the pulse barely reaching the main stream channel (Neal *et al.*, 2003).

2.2 SUPPLY OF THE INDIVIDUAL ELEMENTS

In assessing the risk of nutrient enrichment from a forest site, it is important to know the sources and fractions of nutrients, the structure and chemistry of soils, and to elucidate and quantify the pathways by which water carrying soil and nutrients moves into the aquatic zone.

Low levels of nitrogen and phosphorus enter freshwaters through the natural supply from catchments (Gundersen & Bashkin, 1994; Tunney *et al.*, 1997). In undisturbed catchments, N and P originates from atmospheric deposition and by leaching from soils and rocks (Maberly *et al.*, 2004; Tunney *et al.*, 1997). Undisturbed forests tend to exhibit tight nutrient cycling with only small exports to downstream sites. For example, Poor & McDonnell (2007) observed very low nitrate concentrations and export rates (0.012 kg ha^{-1} $storm^{-1}$, 0.005 kg ha^{-1} $storm^{-1}$, and 0.010 kg ha^{-1} $storm^{-1}$) in an undisturbed forested catchment on moderately deep well-drained silty clay loams and shallow, well-drained silty clays, during storm events in Oak Creek Watershed, Oregon, USA. This was likely due to the lack of anthropogenic inputs in the region. Inputs of nitrate in the forested catchment included atmospheric deposition and nitrogen fixation/microbial processing in the soil, both of which were relatively low locally, resulting in low stream concentrations (Poor & McDonnell, 2007). Natural N and P inputs can be supplemented by anthropogenic sources, including a) fertiliser applied to crop/soil; b) N and P released from decaying organic matter; c) direct inputs of N and P from sewage/organic wastes (e.g. slurry); and d) increased atmospheric deposition.

2.2.1 Fertilisation

The magnitude of the nutrient pressure on receiving waters is related to the percentage of the catchment area that is fertilised. The effect will depend on the soil properties, timing of application, weather conditions, size of the catchment and the characteristics of the water body. The species of tree planted, together with a site's local physical characteristics, determine fertiliser composition and application rates. In Ireland, phosphorus (rock phosphate) is the main nutrient applied, with nitrogen (urea) and potassium (muriate of potash) occasionally applied (Table 2) during the life cycle of the crop, when foliar analysis indicates that there is a nutrient deficiency (Forest Service, 2000a; Jennings *et al.*, 2003). Fertilisers are usually applied manually or mechanically after planting and the requirements differ between species and site. For example, Sitka spruce requirements for phosphorus (granulated rock phosphate, 12.0% Total P) on different sites are based on the following: no application on enclosed and recently farmed land; 250 kg/ha on enclosed and not recently farmed land; 350 kg/ha on unenclosed land (Forest Service, 2000b). The amount of fertiliser applied has decreased steadily over the period 1996-2006, currently at a tenth of the level applied in 1996 (Coillte, *pers comm.*). Where nutrient deficiencies are detected, and branch growth and onsite vegetation prevent the manual application of fertiliser, aerial fertilisation by helicopter can be utilised. Aerial fertilisation is governed by licence by the Forest Service under strict guidelines and exclusion zones that must be adhered too (Anon, 2006). For example, aerial fertilisation must not take place within 100m of the abstraction point of a source of water intended for human consumption or within 50m of an aquatic zone (Anon, 2006). Fertiliser application is also restricted by weather conditions. Application should be avoided during or following heavy rain or if heavy rain is forecast, and in the instance of aerial fertilisation, during high winds (Anon, 2006). To avoid the generally poorer weather conditions of autumn/winter, fertiliser is applied between 1st April and 31st August.

Table 2. The type, concentrations and application rates of fertiliser applied to Irish forests. Adapted from Anon (2006).

Fertiliser Type	*%P*	*%N*	*%K*	*Maximum rate per hectare*
Granulated Rock Phosphate	11-16%	-	-	350 kg
Granulated Urea	-	46%	-	350 kg
Muriate of Potash	-	-	50%	250 kg

In Ireland, fertilistion does not generally occur at planting on reforestation sites. Where it is confirmed by foliar analysis, reforestation sites will be fertilised after 4 years of re-planting (Coillte, *pers comm.*). Urea is also applied to conifer stumps immediately after cutting to prevent disease (Forest Service, 2000b). In 2006, 2.87 tonnes of elemental N was used in the treatment of tree stumps, equating to 16% of the total N used by Coillte in 2006 (Coillte, *pers comm.*).

Fertiliser-induced leaching of phosphorus from peatlands drained for forestry has been the focus of a range of studies in Ireland, Scotland and Scandinavia (Särkkä, 1970; Karsisto & Ravela, 1971; Kenttämies, 1981; Malcom & Cuttle, 1983; Ahti, 1984; Renou & Cummins, 2002; Cummins & Farrell, 2003; Machava *et al.*, 2007), and the results have shown that phosphorus levels in runoff increased after fertilisation. Kenttämies showed an increase of phosphorus concentrations from 18µg l^{-1} of P pre-fertilisation to a mean value of 128µg l^{-1} of P post-fertilisation. Kenttämies estimated that the duration of the leaching of phosphorus fertiliser from a drained peatland could potentially exceed 10 years. Other authors have observed elevated phosphorus concentrations in runoff, associated with fertilisation, after 3 years (Harriman, 1978) and 6 years (Knighton & Stiegler, 1980). In Ireland, studies have also shown that forest fertilisation can have an impact on water quality. Aerial fertilising with rock phosphate (70 kg P ha^{-1} – applied by helicopter on a single day in July 1997) and clearfell harvesting were both associated with increased streamwater concentrations of MRP in the surface water channels observed by Cummins & Farrell (2003). However, in a study on a former agricultural site (Crossmolina, Co. Mayo) that had received regular applications of fertiliser up to 3 years before afforestation, there were no adverse impacts of forest operations on surface water quality. In fact, P concentrations in surface water were actually lower after afforestation than before (Machava *et al.*, 2007). Further results from a phosphorus runoff experiment at Ardvarney, Co. Mayo, showed very high concentrations of P were measured in runoff water. However, the occurrences of the highest concentrations were in the control plots (i.e. no fertiliser treatment). It was suspected that the results obtained were an unanticipated feature of the experimental design (Machava *et al.*, 2007).

2.2.2 Decaying organic matter

A major potential source of nutrient leaching to receiving waters comes via decaying organic matter, including the foliage and branches (brash), unwanted stems, stumps and dead roots (harvest residue), left on site after crop thinning or felling which are added to the soil at the same time that nutrient uptake is reduced. Following thinning, the remaining trees can generally take up these nutrients. However, nutrients which are not retained by vegetation or soil within the catchment will tend to be washed away by overland flow, usually during the first large rainfall event post- harvesting**.** In Ireland, brash is left on infertile sites to increase nutrient supply to the trees of the next rotation. Table 3 shows the nutrient content (N, P and K) of brash from various studies in Europe. It is evident that such brash may contain approximately half the amount of phosphorus recommended in fertiliser prescriptions and double the normal nitrogen application rate (e.g. Hyvönen *et al.*, 2000). Weatherall *et al.* (2006) utilised stable isotope techniques to trace nutrients released by decomposing brash and found that elevated levels of ^{15}N, ^{41}K, ^{26}Mg and ^{44}Ca in new seedlings indicated that nutrients in

Species	*N kg ha^{-1}*	*% of total*	*P kg ha^{-1}*	*% of total*	*K kg ha^{-1}*	*% of total*
Norway Spruce[a]					65-71	53-59
Norway Spruce[b]	280		23			
Scots Pine[a]					82-102	55-66
Scots Pine[b]	74		9			
Sitka Spruce[c]	300	70	31	76	106	73
Sitka Spruce[d]	219		20		71	

brash can make a direct contribution to new tree growth.

Table 3. Nutrient content of brash; % of total = % of total above-ground tree crop at harvest. (adapted from Moffat *et al.*, 2006).
[a] Olsson *et al.* (1996); [b] Hyvönen *et al.* (2000); [c] Stevens *et al.* (1995); [d] Titus & Malcolm (1991).

However, the presence of brash on a site may have negative consequences for the following crop. The uneven distribution of brash on a site can also result in uneven or banded distribution of nutrients and subsequent uneven growth of the crop across the site (Hyvönen *et al.*, 2000). Neal *et al.* (1992) found that the post-felling presence of excessive brash at a site suppressed growth of the subsequent tree crop, which in turn, lowered the rate of nutrient uptake from the soil. The authors recommended rapid establishment of natural ground-layer vegetation following harvesting/land-disturbance to minimise the releases of nitrate to the catchment soils. Fine logging residues, including foliage, fine roots and small branches, are the likely source of leached nutrients soon after cutting because they are the most nutrient-rich parts of the trees (Kubin, 1977) and decompose faster than coarse residues, i.e. stumps, coarse roots and thick branches (Fahey *et al.*, 1988; Hyvönen *et al.*, 2000). The decomposition of coarse residues can take several years or decades because their low concentrations of nutrients and soluble sugars, high C/N and C/P ratios, high lignin concentration and the small surface area to volume ratio associated with large diameter make them resistant to decomposition (Fahey *et al.*, 1991; Hyvönen *et al.*, 2000; Berg & McClaugherty, 2003).

In Scotland, the removal of brash from felled areas decreased nitrogen leaching through elimination of a potential nutrient source and because brash supported mineralisation and nitrification in the soil humus layer (Stevens *et al.*, 1995). Nevertheless, significant elevated leaching following harvesting was found to be usually short-lived (Stevens *et al.*, 1995). The removal of brash could also potentially increase the risk of silt mobilisation, especially on peat and other erodible soils. In Ireland, brash is never removed and measures to conserve or encourage ground vegetation are seen as a more practical and effective way of reducing loss of nitrate-N (Farrell, *pers comm.*). Cummins & Farrell (2003) stated that the P reserves of a site, whether in brash or the soil, may contribute to (although not fully meet) the nutrient demands of the following reforestation crop. Should this residual P contained in brash not be taken into account, post-felling fertiliser application to a site may potentially exceed the sorption capacity of the soil, resulting in run-off risk and potential eutrophication (Cummins & Farrell, 2003). It should be noted that under current Forest Service guidelines this would not occur as foliar analysis would determine if a site required fertilization at re-planting. (Forest Service, 2000b). Historically, the planting of conifer forests in Ireland on peat and other soils with very limited phosphorus retention capacity, combined with pre-Forest Service guideline planting, increased the potential magnitude of the pressure of phosphorus, via decaying organic matter, on Irish waters. However, current practice involves a deliberate crop re-structuring policy by Coillte and compliance with Forest Service Guidelines on harvesting, whereby potential P losses from harvesting are avoided and/or minimised significantly.

2.2.3 Direct inputs from sewage/organic wastes

Municipal and agricultural organic wastes have been identified as the main agents of freshwater pollution in Ireland (Toner *et al.*, 2005). The forest sector is governed by comprehensive legislation and regulations in Ireland (Maguire, 2001), and hence these inputs are negligible in Irish forests.

2.2.4 Increased atmospheric deposition

In Ireland, the spatial maximum deposition of nitrate and ammonium can attain over 20 kg N ha^{-1} $year^{-1}$ in some areas, as opposed to a maximum of 9 kg ha^{-1} $year^{-1}$ for suphur (Aherne and Farrell, 2000). 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). Nitrate leakage from mature forest stands to soil and surface waters has been identified in areas of high nitrogen deposition, including parts of Wales (Stevens *et al.*, 1997) and parts of Ireland (Farrell *et al.*, 2001). In some cases, nitrate leaching to soil and surface waters has been accompanied by elevated concentrations of inorganic aluminium (Farrell *et al.*, 2001).

2.3 PATHWAYS OF PHOSPHORUS AND NITROGEN TRANSPORTATION

Nitrogen and phosphorus have different sources, different routes to a watercourse, and different temporal and spatial dynamics. The whole catchment may potentially contribute nitrate to the drainage waters whereas phosphorus losses often occur from extremely localised regions adjacent to the river channel. It is also apparent that P inputs will display a higher degree of episodicity, compared to N inputs (Pionke *et al.*, 1996). This difference in behaviour of N and P has fundamental implications for assessing their potential biological impact and in identifying appropriate management strategies for reducing nutrient loss at the catchment scale.

P is predominantly lost from catchments through surface run-off during, and post, high rainfall events. The P lost includes the dissolved P (DP)/ SRP fractions present in the original precipitation (typically in the order of 10-100 $\mu g\ l^{-1}$), together with any freshly dissolved or de-sorbed at the soil surface (such as from fertiliser granules) (Reynolds & Davies, 2001). It also includes non-soluble particulate fractions entrained in the flow. Gentry *et al.* (2007) found that concentrations of DRP and PP from tile-drained agricultural watersheds increased with stream discharge and that PP was the dominant form during overland runoff events, which greatly affected the annual TP loads. It has been found that 90% of annual P export from catchments occurs from only 5% of the land area during only one or two storms (Pionke *et al.*, 1997). Phosphorus concentrations and water flows entering and leaving a blanket peat forested catchment pre-, during and post-clearfelling and harvesting operations have been intensely studied in Burrishoole, Co. Mayo (M. Rodgers, *pers comm.*). The average recorded P concentrations in rainfall were 13 $\mu g\ l^{-1}$ total phosphorus (TP) and 4 $\mu g\ l^{-1}$ total reactive phosphorus (TRP), which were similar to the upstream site (14 $\mu g\ l^{-1}$ TP and 6 $\mu g\ l^{-1}$ TRP). At the end of clearfelling the TRP value leaving the forest was 73 $\mu g\ l^{-1}$, increasing to a peak average daily concentration of 187 $\mu g\ l^{-1}$. During the following 7 months (November to May) TRP values decreased to 90 $\mu g\ l^{-1}$. However, after an exceptionally dry summer (June and July), downstream TRP concentrations increased to 500 μg l^{-1}, almost a year after clearfelling and harvesting (M. Rodgers, *pers comm.*). After one year post-clearfell and harvesting, net P release rates were approximately 4000 g TRP ha^{-1} yr^{-1}, in contrast to the average concentrations in the receiving river (5 μg l^{-1} above and 9 μg l^{-1} below the confluence of the study stream and the river). This suggests a dilution effect where the receiving river is diluting the concentration of TRP in the system.

Phosphorus transport in groundwater is generally ignored in catchment studies and monitoring programmes, due to the high P-retention properties of mineral soil (Correl, 1997; Kilroy *et al.*, 1999; cited in Jennings *et al.*, 2003; Reynolds & Davies, 2001). For example, in the 2001-2003 water quality survey of Ireland, phosphate levels in groundwater were not deemed a cause of concern in relation to its use as a drinking water supply (Toner *et al.*, 2005). However, several studies have found that groundwater can be a source of phosphorus to lakes and rivers (Brock *et al.*, 1982; Boar *et al.*, 1995). In Ireland, phosphate in groundwater can contribute to eutrophication of rivers and lakes in some areas, particularly if groundwater provided significant amounts of base-flow during summer months (Toner *et al.*, 2005). Certain groundwaters in the Western River Basin District have been shown to be at risk from increased P levels, where thin soils overlie karst (European Union, 2005b).

The most extensively investigated N cycle disruption is the disturbance by forest harvest. In the classical work of Likens *et al.* (1970) at HBEF, USA, a watershed was clearcut and treated with herbicides to prevent regrowth of vegetation. As plant uptake was inhibited, the nitrate leaching of 142 kg N ha^{-1} $year^{-1}$ constituted an estimation of the maximum nitrification rate in the soil (podzolic soil with $pH < 7$). In disturbed ecosystems where plant regrowth is not inhibited or delayed, the impact of nitrification is much smaller, and the duration of elevated nitrate leaching is only a few years (Vitousek *et al.*, 1979). Disturbances by natural causes (e.g. windfelling, fire) or by thinning may exhibit the same effect on N leaching as forest harvest, depending on the scale of disturbance and the decrease in tree uptake. In Ireland, however, thinning usually involves the removal of about one third of the trees and there should be a sufficient number of remaining trees to absorb N (Forest Service, *pers comm.*).

When assessing the potential loss of nutrients to streams in relation to forestry, there are various pathways to be considered. The amount of supplied phosphorus and nitrogen reaching a stream will be affected by a number of factors including soil type, adsorption capacity of the soil particles, permeability and porosity of the soil, soil disturbance (drainage and ground preparation), water pathways to streams (including drainage patterns), subsurface drainage patterns, vegetation (absorption capacity), rainfall patterns, slope, buffer width and on site buffering capacity.

2.3.1 Soil type (physical and chemical constituents)

The soil acts as a regulator, provided there is no overland flow or flood events, in the relationship between rainfall and runoff, each flow pathway being subject to different obstacles, gradients and storage characteristics (Bell, 2005). Soil characteristics have an important influence on the magnitude of phosphorus losses. In general, sandy and organic soils have a lower capacity to bind phosphorus than those with a high clay content (Sharpley, 1995; Leinweber *et al.*, 1999; Daly *et al.*, 2000). Leinweber *et al.* (1999) found that the risk of phosphorus leaching increased with a decrease in soil clay content. As the capacity for a soil to hold phosphorus decreases, the concentration of phosphorus in soil water increases (Nash & Halliwell, 2000). However, there is the possibilty that clay soils can become saturated, as evidenced in agriculture (Tunney *et al.*, 1997).

Renou and Cummins (2002) presented four case studies examining the influence of forest management operations such as clear-cut harvesting and fertilising on nitrogen and phosphorus concentrations in soil solution and in runoff channels of Irish forests planted on both peat and mineral soils. In case study 1, fertilisation of cutaway peatland forests, MRP concentrations were increased 15-fold in surface run-off water following a manual application of 25kg P/ha of rock phosphate. Although some temporal variability was recorded, the duration of the impact was not known. In case study 2, forest fertilising on peat/mineral mixed soil, MRP concentrations increased significantly the week after the fertiliser was applied but then dropped to pre-fertilisation levels. P levels increased again during the following winter but only in plots which received the highest application rate, i.e. 42 kg P/ha. (superphosphate (16% P) was applied manually in bands, at three different rates (42, 28 and 14kg P/ha)). Heavy showers were recorded during the week following fertilisation. In all other cases, the excess of phosphorus was retained by the mineral soil. However, the high rate of phosphorus fertiliser seemed unsustainable as extra soluble phosphorus was released and leached during the winter. The influence of mineral soil mixed with the peat allows phosphorus to be retained. The timing of application remains critical however, and it is important to avoid high rates of fertilisation. It should be noted that such quick release P fertilisers are not permitted by the Forest Service in routine forest fertilisation.

In case study 3, forest harvesting and fertilising on blanket peatland, the soils of the area were peaty, with no groundwater influence, and had a low nutrient status, and an extremely low phosphorus sorption capacity. P concentrations measured both in stream- and drain-water increased following clear-felling. Post-felling fertilisation also increased P concentrations immediately in both drain water and stream water. Annual median values were well above the Irish thresholds for the eutrophication of rivers. The peak in P concentrations occurred soon after the operation (i.e. felling or fertilising), and the duration of the response was long, at least three years (Renou & Cummins, 2002).

In the final case study, forest harvesting on a mineral soil, the soil was a deep, free-draining mineral soil, with the potential to sorb phosphorus and cations such as ammonium, but it had no ability to retain nitrate other than through uptake by plants. After normal forest harvesting in 1995, decomposition of the foliage residues led to increased availability of nitrogen and phosphorus. The ammonium was rapidly converted to nitrate, and leaching occurred in subsequent months until vegetation became re-established on the site. The degree of destruction of the total root system of the ecosystem was an important factor influencing nitrate loss. Phosphorus leached through the low-sorption organic forest litter layer but was retained by the upper layers of the mineral soil and did not leave the site (Renou & Cummins, 2002).

2.3.2 Rainfall patterns

In Ireland, the average precipitation ranges from approx 750mm per annum on the east coast to over 1200mm on the west coast with the highest annual rainfall recorded in mountainous districts in the west and south-west exceeding 2000mm (pers. comm, Met Éireann). However, a considerable quantity of this precipitation is returned to the atmosphere through evapotranspiration. Schulte *et al.* (2006) described the difference between the total precipitation and the evapotranspiration as the 'net rainfall', which represents the amount of precipitation available for subsurface drainage and overland flow. While Schulte *et al.* (2006) found that the annual amount of 'net rainfall' impacts on nitrate concentrations, phosphorus loss was mainly governed by the occurrence, frequency and timing of intense overland-flow events that followed intense rainstorm events. Within individual overland-flow events, higher rates of flow could be accompanied by increased phosphorus concentrations (Heathwaite & Dils 2000; Kurz *et al.*, 2005).

Results from Schulte *et al.* (2006) suggested that the risk of phosphorus loss was highest in western and northern areas of Ireland, where poorly drained soils carry water surpluses for large parts of the year and supply a pathway for phosphorus loss. In addition, these areas had the highest occurrence of intense drainage systems. Furthermore, their thin, peaty soils and relatively impermeable catchments lead to surface runoff contributing a greater proportion of the supply of water to streams. Also, the high density of lake water bodies in these areas also provide a sink for up to 60% of phosphorus in inflowing rivers which can subsequently be released through climatic action (Kavanagh, *pers comm.*)

In Ireland, river phosphorus concentrations, inferred from 'Q-values' (biotic indices based on running water macroinvertebrate communities, and strongly correlated with median P-levels (McGarrigle *et al.*, 2002)), were found to be highest in areas with the highest livestock stocking densities (McGarrigle *et al.*, 2002; Schulte *et al.*, 2006). The authors suggested that, although the pathways for phosphorus loss were more prevalent in northern and western areas, the absence of significant pressures in these areas explains the generally good river water quality in this region. However, it could also reflect that forestry pressure has not yet manifested, as trees have only recently reached felling age, and the potential risk could still be high.

2.3.3 Water pathways to streams

Rainfall falling on a site reaches receiving waters through four main pathways: a) surface run-off, b) soil throughflow c) percolation into the groundwater and d) via drains/channels. In a given catchment, the contribution of water from the different routes to a stream will depend on the soil characteristics and the underlying rock type, slope, rainfall patterns and the intensity of land management.

Surface run-off tends to occur more frequently on poorly-draining soils such as peat or heavy clays or on very thin soils over bedrock or iron pans. It is most evident during heavy rainfall, or after prolonged rain resulting in saturated soils. Throughflow is associated with more permeable soils, e.g. brown earths and brown podsolic soils. Groundwater percolation is only significant in very freely draining geology, such as limestone. Slope will determine the speed of runoff, such that steep slopes will tend to favour surface runoff over throughflow. High intensity rainfall events will also tend to favour surface runoff, through rapid saturation of surface soils (Nihlgård *et al.*, 1994). Artificial drainage channels (for the crop and for road building) effectively increases surface runoff and tends to channel the water flow.

2.3.4 Vegetation, (catchment and riparian)

Vegetation mediates the transport of N and P through a catchment in two ways; a) by tightly re-cycling the nutrients within the catchment; and b) by mediating the influence of rainfall on soil and throughflow of water. Due to the ability of trees to trap nutrients, nutrient levels are generally relatively low in streams draining mature forests (Neal *et al.*, 1992). After clearfelling, nitrate levels in stream water can rise rapidly but have been shown to decline relatively quickly following soil stabilisation and regrowth of vegetation with nitrates generally returning to pre-felling levels within three years (Martin *et al.*, 2000).

2.3.5 Slope

Steeper slopes lead to greater soil erosion and more rapid nutrient loss (e.g. Johnson *et al.*, 2000). Surface runoff is faster on a steep slope, resulting in less contact between water and the soil and vegetation, and less time for sediment to settle out or nutrients to be taken up. The Forestry and Water Quality Guidelines allocate slopes into the following categories (Forest Service, 2000a): < 15% - moderate slope, 15-30% - steep slope, > 30% - very steep slope. The slope and sensitivity to erosion is used to determine the buffer zone width, which varies from 10m (minimum buffer width) to 25m.

Hill slope and catchment scale studies are necessary to trace the fate of P once mobilised along the pathway of surface runoff. This is because it is still not clear to what extent particular P fractions, once mobilised, are modified before they reach the stream or, where the P source area is some distance from the stream, whether mobilised P actually reaches it. A recent catchment wide study in Ireland has shown a net contribution of 3% to P loading from forestry (Lough Leane, 2002). Also, studies of P losses from experimental plots in Ardvarney, Co. Mayo (Machava *et al.*, 2007). have demonstrated high levels of P in surface runoff (means ranging from of 0.491 mg l^{-1} to 1.426 mg l^{-1}).

2.4 RECEPTOR SENSITIVITY

Waters vary in their sensitivity to nutrient enrichment from forestry. Upland waters that are naturally nutrient-poor (oligotrophic) and where biological activity is usually phosphorus-limited may be particularly sensitive (Mulqueen *et al.*, 2004). In extreme cases, P and N enrichment can produce excessive algal growths, resulting in dissolved oxygen fluctuations and disruption of the ecosystem (Correll, 1997; Chambers *et al.*, 2006).

2.4.1 Primary production

P has an impact on stream systems via the biofilm/macrophytes. P on its own is rarely important, although it may be toxic in some cases. N, however, can be directly toxic, in the form of ammonia and nitrite. The uptake of P and N by biofilm/macrophytes is determined by the action of other limiting factors on algal growth in upland waters such as low temperature, light and silica levels (Nisbet, 2001)

The forest surrounding most small streams provides heavy shade which usually limits primary production. At Coweeta (US), periphyton production was typically low with estimates of about 3 kg (dry weight) m^{-2} $year^{-1}$ in streams draining undisturbed forests (Webster *et al.*, 1983). However, canopy removal associated with logging increased solar insolation of the stream and periphyton production increased to 84 g m^{-2} $year^{-1}$ which provides a potential sink for NO_3^- and P.

Phytoplankton only forms a significant component of slow-moving rivers in lowland regions where the retention time of water is longer than the generation time of the plankton (Lampert & Sommer, 1997). As well as flow rate, turbidity is a critical factor in limiting the development of a phytoplankton community in rivers (Skidmore *et al.*, 1998). Monitoring data spanning 21 years from the River Nene in England, showed a weak relationship between phytoplankton abundance and nutrient conditions, with stronger relationships found in spring (Balbi, 2000). Discharge, temperature and light were significant predictors of spring chlorophyll a concentrations.

The phytobenthos, or benthic and periphytic algae, are considered to be the main source of primary production in many streams and rivers (Wetzel, 2001). As with river phytoplankton, periphyton biomass in running water is not generally strongly related to nutrient concentrations. It is more often most strongly affected by other stress and disturbance factors, such as current velocities, frequency of floods, turbidity and light availability, grazing and substratum type (Welch *et al.*, 1992). In general, however, numerous studies have demonstrated that nutrient enrichment in the overlying water usually results in enhanced growth of benthic algae. More specifically, limitation by nutrients is most frequently observed during summer months (Francoeur *et al.*, 1999; Biggs *et al.*, 1998; Stanley *et al.*, 1990).

Measures of phytobenthos abundance are, therefore, not simply a response to ambient water column nutrient concentrations. Algal production in thick mats is maintained more by efficient internal recycling of nutrients than by external diffusion (Wetzel, 2001). The role of the substrate as a source of nutrients for the phytobenthos community is particularly marked in oligotrophic waters (Wetzel, 2001).

Limited studies exist on the importance of nutrient limitation with respect to river macrophytes. Although there have been numerous studies examining the relationship between aquatic plant communities of running waters with various environmental variables (e.g. Demars & Harper, 1998), few, if any, have been able to identify clearly the effects of nutrients on macrophyte communities in a single catchment, because of the synergistic effects of other physical, chemical and biological factors. Wilby *et al.* (1998) found macrophyte cover in running waters to be influenced by factors such as flow, water temperature and solar radiation. Other influencing factors include sediment type, turbidity, water depth and grazing (Flynn *et al.*, 2002).

The effect of forest fertilisation on water supply reservoirs is of particular concern in Scotland where drinking water is taken from upland lochs and reservoirs. Excessive algal production resulting from P runoff can lead to financial and technical problems associated with clogging of filters at treatment works. The development of certain species of blue-green algae with toxic by-products would be more serious (Best, 1994).

2.4.2 Macroinvertebrates

Nutrient enrichment may adversely effect stream animal communities. Enriched streams have increased invertebrate biomas and altered invertebrate communities (Bourassa & Cattaneo, 1998). Community structure has been correlated directly with P concentration (Miltner & Rankin, 1998). Miltner and Rankin (1998) found that the biotic integrity (the species composition, diversity and functional organisation of a community, compared to that expected for the natural habitat of a region (Karr & Dudley, 1981)) of rivers and streams was negatively correlated with increasing nutrient concentration, particularly phosphorus. In extreme cases, levels of primary production can be stimulated by nutrients; organic C will build up in the system and cause a subsequent low dissolved oxygen and high pH event. As a result, fish and invertebrates will grow poorly and even die if the oxygen depletion and pH increases are severe (Welch, 1992).

An increase in primary productivity after fertiliser addition has a knock-on effect on secondary production. For example, higher densities of filter-feeding invertebrates such as Simuliidae (blackfly) larvae can result (Best, 1994). Invertebrate grazing can result in reduced chlorophyll yield per unit nutrient in streams regardless of regional differences (Bourassa & Cattaneo, 1998), as is the case with lakes.

2.4.3 Fish

Salmonids, particularly the younger life stages, have a relatively weak tolerance to low oxygen concentrations and are thus particularly vulnerable to organic pollution (Eklov *et al.*, 1999; Crisp, 2000; Armstrong *et al.*, 2003). Decaying organic matter in bed sediments can have a very severe impact on eggs and alevins through anoxia. In nitrate-rich rivers, denitrification occurring in anoxic, silted sediments can rise concentrations of nitrites and ammonia leading to increased mortality of brown trout egg and embryos (Massa *et al.*, 2000). In fact, the increased mortality of eggs and alevins in anoxic sediments is considered to be the single biggest problem for trout populations in streams flowing through agricultural catchments (Bagliniere & Maisse, 2002). Levels of > 1mg l^{-1} NH_3-N in Ohio streams had negative impacts on the fish communities (Miltner & Rankin, 1998). A loss in the number of sensitive fish species, decreased relative abundance of top carnivores and insectivores, and an increasing proportion of tolerant or omnivorous fishes were correlated with nutrient enrichment (Miltner & Rankin, 1998).

Although there are many accounts of salmonid declines in eutrophicated waters, there has been little investigation into the role of moderate enrichment of streams by diffuse nutrient inputs, at levels too low to causes serious harm to ecosystem health. At these levels, fish production or biomass in rivers has been shown to be positively related to phosphorous concentration, indicating strong trophic links between primary producers and consumers (Hoyer & Cranfield, 1991; Randall *et al.*, 1995). Invertebrate secondary productivity is in turn often limited by nutrient supply (Huryn & Wallace, 2000). Several authors have reported artificial nutrient enrichment of streams leading to enhanced fish populations (Johnston *et al.*, 1990; Perrin & Richardson, 1997; Deegan *et al.*, 1997; Slavik *et al.*, 2004). Instream food availability is one of the major factors (along with hydraulic forces and temperature) controlling the growth, fecundity and production of salmonids (Bagenal, 1969; Egglishaw & Shackley, 1985). Enrichment of streams can thus lead to high growth rates of salmonids through high abundance of suitable food items (Gibson & Colbo, 2000) and/or competitive displacement of weakly competitive species by dominants (Taniguchi *et al.*, 2002).

2.4.4 Forest operations

Cummins and Farrell (2003) reported strong seasonality in MRP concentrations in two streams following combined felling and fertilising in Cloosh Valley Forest, Co. Galway. Over the period from January 1996 to December 2000, both harvesting and fertilisation were followed by an immediate rise in MRP concentrations, with subsequent and generally exponential declines of MRP, independent of season. However, in one stream with natural drainage, an annual cycle of a late summer maximum and a spring minimum occurred.

Unless precautions are taken while forest operations such as site preparation and harvesting are being carried out, soil disturbance can lead to increased concentrations of total suspended solids (TSS) in surface water, which is usually the main source of particulate P in watercourses (Nour *et al.*, 2006). Sediment concentrations and water flows entering and leaving a blanket peat forested catchment pre-, during and post-clearfelling and harvesting operations has been intensely studied in Burrishoole, Co. Mayo (M. Rodgers, *pers comm.*). Suspended sediment (SS) concentrations upstream of the intact forested catchment ranged from 0 to 5 mg l^{-1} during base flow conditions and from 5 to 31 mg l^{-1} during flood events. Downstream, the SS concentrations during base flow conditions pre-, during and post-clearfelling ranged from 0 to 5 mg l^{-1}. During the 2 month period of clearfelling and harvesting, approximately 13 kg SS ha^{-1} was released from the intact forest catchment and approximately 45 kg SS ha^{-1} from the harvested catchment. Understanding the transport of P from the afforested area to aquatic ecosystems is essential to the development of methods to minimize the potential export of phosphorus from forest ecosystems. In the first year after clearfelling (2012 mm rainfall), the net SS release rate from the harvested catchment was 450 kg SS $ha^{-1}yr^{-1}$ in comparison to 172 kg SS $ha^{-1}yr^{-1}$ from the undisturbed forest catchment (M. Rodgers, *pers comm.*)

Few studies have investigated the impacts of afforestation on eutrophication in Ireland. The most recent study was by Machava *et al.* (2007) who studied an ex-agricultural site that had been extensively drained and fertilised, and where water quality was examined before forest establishment, during establishment operations and for a short period following establishment. Sampling commenced in September 2002 and ceased in June 2004 (site preparation and planting occurred in March/April 2003). There was no indication of an adverse effect on water quality resulting from the forest operations. In fact, although pipe drains on the site showed very high phosphate concentrations before the study began, measurements at the main outlet taken during the study were negligible. It was found that concentrations were actually lower at the end of the study period than prior to establishment. It was believed that this may have been due to a lowering of the water table due to improved drainage and reduced surface runoff, so much so that the water table was below the zone of P in the soil and hence resulted in no leaching of P (Machava *et al.*, 2007).

The impacts of afforestation on water quality was studied in the headwaters of the River Clydagh between 1991-1997 as part of an ongoing environmental impact assessment (Giller *et al.*, 1991,1993,1994,1996; O'Halloran *et al.*, 1995,1997). A baseline survey was instigated in 1991 before site preparation and planting was undertaken in 1992. Sampling was carried out for each subsequent year to a final survey in 1997, studying the impacts of afforestation (and associated operations e.g. aerial fertilisation) on aquatic flora and fauna and water chemistry. From the time of planting there was an initial increase in plant cover and the extent and distribution of *Cladophora* until the 1995 re-survey where the trend reversed (although this did not relate to a decrease in nutrient levels, indeed nutrient levels had increased from previous surveys). In 1996 there was continued decrease in the extent and distribution of *Cladophora*, accompanied by a decrease in nutrient levels. In the final survey, the abundance and distribution of *Cladophora* returned (although the levels were not to the same extent). This apparent increase in algae was not matched by any evidence of an increase in surface water nutrients. It was suggested that these algae were possibly "sinking" the water nutrients through plant growth and that some of the nutrients for plant growth may well be from the nutrients associated with the evident peaty siltation (O'Halloran *et al.*, 1997).

From the initial survey to 1995 there was a continued decrease in some sensitive macroinvertebrate taxa, an increase in and dominance of tolerant forms (e.g.

chironomidae) and a slight decrease in water quality ratings that indicated a degree of deterioration throughout the study area except for the most upstream site above the forest. These changes supported the similar trends in water chemistry and plant data and confirmed the conclusions that the River Clydagh was suffering from a degree of enrichment since the new forestry operations began. Increases in siltation levels were also believed to partly explain these changes (O'Halloran *et al.*, 1997).

Over the last two years of the survey the macroinvertebrate community stabilised to a degree with overall densities recovering somewhat from the low 1995 levels. Taxon richness was generally higher than it had been while diversity values were similar to 1993 levels, but several sites still showed strong dominance either by chironomid larvae or by *Baetis rhodani*. Water quality ratings were largely unchanged compared to 1995 and were still lower than in 1991. In general, although the macroinvertebrate community had stabilised somewhat, it had not recovered since the disturbances associated with development in 1991, and the loss or more restricted distribution of several sensitive invertebrate species over the past 5 years was of some concern.

The data from that study suggested that the continued influence of afforestation on the water quality of the River Clydagh may be through enrichment (although the trend for increased nutrient levels appeared to have ceased) or some physical changes, particularly increased sediments and particulate organic matter rather than through alterations in pH and alkalinity as found elsewhere (UKAWRG, 1994; Cruikshanks *et al.*, 2006). Aerial phosphate fertiliser applications could present a significant threat and must be carefully planned to ensure that phosphate losses from potential additional applications in a given catchment do not exceed environmental quality standards in receiving lakes or reservoirs.

The Department of the Environment and Local Government initiated two studies, namely Lough Leane (Lough Leane, 2002) and the Three Rivers Project (Three Rivers Project, 2002) in catchments with forestry together with other land practices, such as sheep grazing, to estimate the contributions of forest operations, e.g. harvesting, forest cultivation and fertilisation, to the overall P loading from forested land. The findings from both studies indicated that high flow rather than high nutrient concentrations from forest operations was the primary determinant of water quality in each study area. The results of the Lough Leane report showed that agriculture accounted for the majority of the phosphorus input (47%), with forestry contributing a 3% input (Lough Leane, 2002). In response to concerns expressed by the Lough Leane Working Group in April 1999, Coillte reduced its 1999 aerial fertilisation programme in the Clydagh Valley, from 4.8 tonnes of elemental phosphorus to 1.3 tonnes of elemental phosphorus. Water monitoring results showed that although elevated levels of soluble phosphorus were observed in forest drains located in the fertilised compartments, little to no significant change in water quality was detected in adjacent tributary streams and the Clydagh River after fertilisation (Lough Leane, 2002).

Many international studies have shown the release and leaching of nutrients, particularly nitrates, following clearfelling into stream waters (Bormann *et al.*, 1974; Neal *et al.*, 1992; Jones *et al.*, 1998; Martin *et al.*, 2000; Johnson, 2006). Stevens *et al.* (1995) found increases in nitrate export into streams following felling, particularly in association with soil run-off. It was suggested that the source of nitrate was from the death of fine tree roots, followed by rapid mineralization and nitrification (Stevens *et al.*, 1995). In the same study one-third of the P (10 out of 30 kg ha^{-1}) leached out of the brash lying on the clearfelled plots within one year of felling and was immobilized in the soil (Stevens *et al.*, 1995).

In the UK a major response to felling is an increase in nitrates (Neal *et al.*, 1992; Neal *et al.*, 2003; Neal *et al.*, 2004). Also, the disturbances in water quality associated with felling ebb after a few years as nitrate uptake by tree re-establishment and growth increases and the concentrations of the major acid-anions decrease. Indeed, this can even lead to water quality better than in the pre-felled condition (Neal & Reynolds, 1998; cited in Neal *et al.*, 2004).

3. PRESSURE - SEDIMENTATION

Suspended solids are insoluble particles that either float on the surface or are in suspension in the water, and tend to increase with increases in river flow as a result of scouring of the river bed and banks (Naden *et al.*, 2003). Silt and clay will normally travel in suspension while sand may be transported either in suspension or as bedload dependent on the flow characteristics. It has been shown in the UK that 95% of suspended solids sampled in rivers are usually < 0.063mm in size (Walling *et al.*, 2000).

Sedimentation can be defined as the deposition of fine sediment either within a gravel substrate or lying on the surface of the stream bed (Naden *et al.*, 2003). The deposition of sediment on the surface will occur as a result of settling whenever or wherever the upward momentum transfer from the fluid motions is reduced below the weight of the suspended solids (Naden *et al.*, 2003). This can occur, for example, near channel margins where the velocity is reduced (Tipping *et al.*, 1993), surface layers within pools (Lisle & Hilton, 1999), and around vegetation (Sand-Jensen, 1998). This produces a diversity of sediment patches over the riverbed, which is essential for good habitat diversity and to the different life stages of some species. Problems arise via high deposition rates, smothering of coarser patches with finer sediments, and finer materials that may reduce oxygen levels either through a decline in throughflow rates or, in the case of organic particulates, by their own use of oxygen (Naden *et al.*, 2003).

Riverbed sediments can play a significant role in buffering concentrations of soluble reactive phosphorus (SRP) in surface waters (House & Denison 1998, House & Warwick 1999). This is particularly evident under low-flow conditions, where water residence times are high (and there is a relatively long contact time between the water column and the bed sediment) and where the sediment surface area to water volume is high. However, this is unlikely to occur in many Irish forested streams, which have naturally high flow rates and few fine sediments.

Substrate condition is a function of existing and available sediment, flow regime and the detailed topography and hydraulics of the site. To fully grasp the response of an individual site, an understanding of the sediment dynamics of the catchment and upstream river system is necessary, as sediment may be derived from either in-channel (bed and bank) or catchment sources (Naden *et al.*, 2003).

3.1 SOURCES OF THE PRESSURE

3.1.1 Sediment Sources in Forestry

The main potential sources of sediment from forests are as follows; a) disruption of the soil surface, causing the subsoil to be exposed to erosion and eventually, the transportation of the finer particles by overland flow; b) weathering of parent material resulting in particle movement by overland flow (parent material comprising non-consolidated material represents a high risk); c) the transportation of loose or decaying organic particles; and d) drainage systems, which reduce the length of overland flow paths and, thereby, the settlement of sediment on land. Drains can also potentially be a source of sediment by means of erosion.

The risk of soil erosion arises where the soil is disturbed or exposed (i.e. little or no vegetation). Erosion is a process caused by the action of wind, water and gravity on soils with silt being the most easily eroded sediment size (Graf, 1971), as it is smaller than sand and yet not small enough to experience significant cohesive forces. However, the vulnerability of soils to erosion depends not only on such factors as climate, topography and soil characteristics, but also upon the type and intensity of land use (Carling *et al*., 2001). Forestry is one such land use that can potentially contribute to increases in sedimentation in water systems (e.g. Giller *et al*., 2002).

3.1.2 Direct Effects of Forestry Activities

Soil is disturbed and exposed at various stages of the forest life cycle, the most important being site preparation, drainage, road/bridge construction and harvesting. Soil erosion can continue to occur as long as there is exposed soil on site. In practice, this means that erosion can occur until the soil has been stabilised by vegetation cover (Horswell & Quinn, 2003). The sediment issue is the major water quality concern in relation to forestry in other parts of the world. For example, in their comprehensive review of forest practices as non-point sources of water pollution in North America, Binkley and Brown (1993) noted that concentrations of suspended sediment often increased after road construction, harvesting and site preparation. However, the response to forest practices depended strongly on site and weather conditions and whether appropriate ameliorative measures had been adopted to minimise potential impacts.

The key factor in determining the quantity of sediments released, relevant to all stages of forest operations (e.g. site preparation, road construction, and harvesting), is the size of the working zone (e.g. area or length of road) in which soil disturbance takes place.

a) Site preparation

Ground preparation invariably involves some measure of soil disturbance and therefore has a direct influence on water quality (Forest Service, 2000b). In general, when site preparation increases the exposure of bare soil and removes more vegetation, it accentuates water quality problems, especially where sufficient slopes exist (≥ 15%) (Aust & Blinn, 2004). The magnitude of the pressure of sedimentation at this stage depends on the size of the plantation, soil type, slope and the technique used for ground preparation. In Ireland, mounding is the most common type of soil preparation used, as it suits wet mineral soils (Forest Service, 2000b). Mounding, which involves the creation of open drains, thus lowering the water table, and the distribution of soil for planting, allows variation in drainage and can help to establish stable growing conditions. It is effective on sites which have excessive soil moisture, low soil temperatures, subject to frost or strong vegetation competition. Mounding also helps soil aeration but, in dry conditions, can give rise to loss of soil water (Forest Service, 2000b).

Ground preparation (ploughing and development of associated drainage systems) in upland areas has a marked effect on the timing and quantity of runoff, with peak discharges increasing and the duration of storm hydrographs shortening which results in increased soil erosion (Gee & Stoner, 1988). Since the 1980s, ground preparation for forest establishment in Ireland has moved away from ploughing in favour of more sensitive cultivation techniques such as mounding to minimise the risk of sedimentation. Machava *et al.* (2007) found that site preparation, on a cambic podzol site previously used for agriculture, produced no sustained increase in total suspended solid (TSS) concentrations, with a mean TSS concentration of 5.57 mg l^{-1} in the main outlet stream. Sites that have been ploughed in the UK have shown dramatic levels of suspended solids entering waterbodies. For example, in Northumberland, the TSS concentration was typically >200 mg l^{-1}, five years after the initial work was undertaken, compared to pre-ploughing values of approximately 10 mg l^{-1} (Best, 1994). Elsewhere, the peak concentration of suspended solids in the drainage water entering a feeder river of Loch Lomond in Scotland was as high as 1200 mg l^{-1} (Best, 1994). This suggests that site preparation utilising mounding reduces the potential of sediment loss, although further studies need to be carried out to clarify this.

b) Drainage

The magnitude of the pressure of sedimentation at this stage depends on the extent of the area drained, soil type, slope and the technique used for drain construction. As mentioned previously, mounding is the most common practice of ground preparation in Ireland. This process incorporates drainage which lowers the water table and also provides a suitable medium on which to plant a sapling by the placing of the excavated material from the newly created drain in mounds at 2m by 2m spacing on the site. As well as for allowing establishment, mounding also raises the sapling above the surrounding ground vegetation. This in turn helps to reduce the need for weed control at later stages. In some soil types (e.g. peaty soils) drainage favours surface pathways and lead to faster runoff so that streams peak earlier and reach higher flow rates (Owens *et al.*, 2005). In these circumstances, soil disturbance can result in a marked increase in erosion and sedimentation of nearby freshwaters, particularly if the operations are followed by prolonged and heavy rainfall. Faster run-off could also reduce the time that conditions remain suitable for fishing and fish migration, contribute to downstream flooding, and enhance the erosion of stream channels, leading to loss of habitat and land. The Forestry and Water Quality Guidelines (Forest Service, 2000a) provide guidance on good drainage design such as collector drains excavated at an acute angle to the contour (0.3%-3% gradient) to minimise flow velocities, and drainage channels that taper out before entering the buffer zone to ensure that discharged water infiltrates out over the buffer zone.

c) Road construction

The construction of forest roads can contribute significantly to soil erosion particularly where planting has occurred on steep slopes and in forestry establishment and management practices (Brogan *et al.*, 2002). Sediment production from forest roads has been highlighted as the major source of the increased sediment loads often resulting from harvesting operations (Burroughs *et al.*, 1991). Slope destabilisation leading to mass soil movement, the wash-out of roads due to undersized culverts, the erosion of steep road drains, embankments and cuttings, and the wash-off of fine sediment from road surfaces during use of forestry traffic, have each been responsible for damaging the freshwater environment (Burroughs *et al.*, 1991). The pathway susceptibility for road construction therefore tends to be direct.

The magnitude of the pressure of sedimentation at this stage depends on the type and extent of the construction, the positioning of the construction, and the associated techniques used. The use of unsurfaced roads can rut and erode the road directly, or make it more susceptible to frost action and rainfall-runoff induced degradation (Carling *et al*., 2001). Changes in groundwater hydrology owing to ill-positioned roads can lead to hillslope instability and landslides on vulnerable soils and subsoil materials (Carling *et al*., 2001).

A major disturbance during timber harvesting is road construction. Substantial increases in sediment yields have been noted on watersheds during and following the construction of forest roads. Erosion rates on roads and landings in southwestern Oregon were 100 times those on undisturbed areas, while erosion on harvested areas were 7 times that on undisturbed sites (Amaranthus *et al.*, 1985). However, in Ireland, road construction takes place prior to harvesting and never during harvesting.

Sheridan and Noske (2007) found a very strong relationship between the amount of traffic on forest roads and sediment delivery. They produced a model incorporating factors of annual rainfall and road slope to predict annual sediment loadings from gravel-surfaced forest roads. The tolerance of in-stream organisms to specific patterns (intensity, frequency, and duration) of sediment input pulses will determine the net impact of road runoff on the local aquatic ecosystem. Factors that Sheridan and Noske (2007) found that should be taken into consideration when prioritising remedial work were rainfall, traffic loads, road slope, soil type, surfacing, and road catchment area.

d) Harvesting

The potential transfer of sediment as a result of harvesting is dependent on the percentage of the plantation harvested and the ground slope, so the size of the clearfell area is a key factor in determining the quantity of sediments released (Johnson *et al.*, 2000). Reducing the portion of the catchment to be felled at any one time reduces the amount of sediment lost as the relative area of exposed soil is smaller. The risk of sedimentation from thinning operations is generally less than that from clearfelling.

In a review on forestry practices and water quality in the eastern United States, Aust & Blinn (2004) concluded that the increased soil disturbance and water movement caused by timber harvesting results in slight, but measurable increases in stream sediment and nutrients. As vegetation recovers, the transpiration increases and bare soil is covered. This was found to occur 2 to 5 years following harvest, and resulted in water quantity and quality recovery. Most water quality problems associated with forest harvesting were caused by poorly designed and constructed roads and skid trails, inadequate closure of roads and skid trails, stream crossings, excessive exposure of bare soil, and lack of adequate buffer zones (Aust & Blinn, 2004).

The type of machine utilised for clearfelling can affect the level of soil disturbance. Impacts of clearfelling in the USA showed that 15% of logged areas suffered deep soil disturbance as a result of tractor logging whereas cable logging produced the same degree of disturbance on only 1.9% of the area (Van Hook *et al.*, 1982).

3.2 SEDIMENT TRANSPORTATION

In assessing the risk of sedimentation from a site, it is important to understand the pathway by which water carrying soil and sediments moves into freshwaters. The vulnerability of a site to erosion, resulting in potential sedimentation, can be assessed based on the following relevant factors.

3.2.1 Soil type

Certain soils (e.g. peaty soils) are more erodable than others. Soils with a high clay content (sticky soils, e.g. gleys) are less likely to erode than those that are crumbly in nature (e.g. soils derived from Old Red Sandstone). The extent to which soil will erode is related to slope and runoff.

3.2.2 Slope

In general, the steeper the slope the greater will be the likelihood of soil erosion. Steep slopes lead to rapid surface run-off and fast flow in drains, and represent the highest risk of sedimentation (Johnson *et al.*, 2000). Rivers and streams can then transport sediments downstream as either suspended solids or as part of the bed load. Altitude remains the most important variable in explaining the rainfall pattern, at least at ground level, although slope is important (Kirby *et al.*, 1991). In the four months before the forestry drainage, suspended sediment concentrations were very low and averaged less than 4 mg l^{-1}, despite samples being taken over a wide range of discharge conditions. The annual sediment load was estimated to be about 3t km^{-2} - generally, higher annual yields were recorded at other upland sites (Robinson *et al.*, 1998). Factors limiting sediment yields at Coalburn included the generally gentle slopes and armouring of the channel bed by cobbles (Robinson and Blyth, 1982).

3.2.3 Physical structure/type of vegetation

This determines the efficacy of sediment entrapment. For example, flow through tall, dense grasses should reduce flow speed and total volume of flow through infiltration, which in turn is likely to lead to deposition of sediments within the vegetation (Carling *et al.*, 2001). Leaf type is likely to influence sedimentation. Sediments will tend to be trapped more effectively by vegetation with broad/hairy leaves than vegetation with narrow/smooth leaves (Brown & Brookes, 1997; cited in Carling *et al.*, 2001).

3.2.4 Climate

Erosion is linked to climatic severity, which varies with altitude and exposure. Rainfall duration and intensity will affect erosion. Soil particles move during and immediately after heavy rainfall, with heavier rainfall resulting in the movement of larger soil particles. If rainfall runs off the soil surface or through drains, there is a risk of sedimentation. If the water percolates into the soil, the risk of sedimentation is low (Horswell & Quinn, 2003).

3.3 RECEPTOR SENSITIVITY

3.3.1 Risk Assessment

The deleterious effects of high suspended solid loads and sedimentation on riverine habitats have been well documented (e.g. Wood & Armitage, 1997). Concern has focused on the erosion and siltation resulting from cultivation, harvesting, drainage and road building operations in relation to forestry (Leeks & Roberts, 1987) as these processes can have detrimental effects on both macroinvertebrates and salmonid spawning sites due to sediment smothering the streambed and decreasing oxygen levels. Increases in the amount of soil delivered to the stream can thus greatly impair, or even eliminate, fish and macroinvertebrate habitat and change the structure of the physical habitat. Large inputs of coarse sediment can have a serious impact, leading to the destabilisation of stream beds and channels, the shallowing of watercourses, blockage of pipelines and water intakes to treatment works, and a long-term reduction in reservoir storage capacity.

It is therefore imperative to be able to identify sites that are potentially at risk from suspended solids. The risk assessment methodology utilised in Ireland to identify these sites is based on areas of high erosion potential identified using peat soil and sandstone derived soil layers' from the Teagasc National Soil Maps, a generated critical slope map (critical slope is >= 15% slope) and 60m buffer (either side) of river water bodies, and extracting critical young forestry types from the Forestry Inventory and Planning System (FIPS) database in conjunction with EPA records of river SS scores and river Q data (European Union, 2004). The impact potential contains factors that lead to the creation of a potential for erosion, namely vulnerable soils, steep slopes, proximity to watercourses and presence of coniferous forest.

3.3.2 Impacts on Aquatic Biota

High turbidity levels due to inputs of fine sediments such as clay, silt and fine sand can have an adverse impact on the aquatic flora and fauna.

a) Primary Production

The impact of sedimentation on producers in streams and rivers has extensive consequences since periphyton and aquatic macrophytes form the base of the food chain and any adverse impacts will probably also be evident in the invertebrate and fish communities. Fine sediment suspension and deposition affects producers in four main ways:

(1) By reducing the penetration of light and, as a result, reducing photosynthesis and primary productivity within the stream (Wood & Armitage, 1997).

(2) By reducing the organic content of periphyton cells (Cline *et al.*, 1982; Graham, 1990).

(3) By damaging macrophyte leaves and stems due to abrasion (Lewis, 1973a,b).

(4) By preventing attachment to the substrate of algal cells, and by smothering and eliminating periphyton and aquatic macrophytes in extreme instances (Brookes, 1986).

Aquatic macrophyte growth has important implications for the hydraulic conditions within a stream. Seasonal growth of both marginal and instream macrophytes influences flow velocity and secondary flow patterns, creating areas of slow and fast flowing water, increasing channel roughness and water depth, and increasing habitat diversity (Wood & Armitage, 1997). Macrophyte stands can therefore enhance the deposition and accumulation of fine sediments (Carpenter & Lodge, 1986), trapping sediment particles that settle out and are deposited beneath them.

During a study in the south west of Ireland (Gallagher *et al.*, 2000; Johnson *et al.*, 2000) it was found that although clearfelling caused an increase in canopy openness at sampling locations where a buffer strip was absent, increases in green algae and macroalgae did not always result. This was suggested to be as a result of a decrease in the penetration of sunlight into the water column due to increased levels of suspended sediment, which delayed or prevented algae growth (Gallagher *et al.*, 2000).

b) Macroinvertebrates

Fine sediment suspension and deposition affects benthic invertebrates in four ways: a) by altering substrate composition and changing the suitability of the substrate for some taxa (Vuori & Joensuu, 1996); b) by increasing drift due to sediment deposition or substrate instability (Suren & Jowett, 2001); (3) by affecting respiration due to the deposition of silt on respiration structures or low oxygen concentrations associated with silt deposits (Lemly, 1982); and (4) by affecting feeding activities by impeding filter feeding due to an increase in suspended sediment concentrations (Aldridge *et al.*, 1987), reducing the food value of periphyton (Cline *et al.*, 1982, Graham, 1990) and reducing the density of prey items (Peckarsky, 1984; cited in Wood & Armitage, 1997).

Although sediment-caused decreases in invertebrates are most often attributed to reduction in habitat availability, abrasion and clogging of respiratory systems by suspended particles also can be detrimental, especially to filter-feeding functional groups such as Trichoptera (Garman & Moring, 1987).

In Ireland, Johnson *et al.* (2000) found negative impacts on the macroinvertebrate community, such as reduced taxa richness, particularly of mayflies, stoneflies and caddisflies, at sites that had been subject to relatively large and longer term inputs of soil and suspended solids. However, rather than being associated with direct run-off, slippages, drainage channels or bank collapses, most of these sites were located downstream of crossing points for machinery where no logs or other materials had been used to cross the stream and where relatively large and long-term inputs of suspended solids, sediment/soil occurred (Johnson *et al.*, 2000).

An increase in the volume of fine sediments favours some benthic invertebrates at the expense of others. Some taxa, such as Chironomidae, utilise fine sediments in the construction of cases and tubes (Dudgeon, 1994; cited in Wood & Armitage, 1997), and

Oligochaeta and Sphaeriidae are frequently associated with fine sediment (Armitage, 1995; cited in Wood & Armitage, 1997). However, there have been relatively few studies on the effects of fine sediment deposition on individual taxa.

c) Fish

Suspended solids can have both direct and indirect effects on fish populations. The effects of fine particle suspension and deposition on fish are better documented than for other organisms. At least five ways in which high concentrations of fine sediment adversely affect lotic fisheries have been identified:

(1) By adversely acting on the fish swimming in the water and either reducing their rate of growth, reducing their tolerance to disease or killing them; lethal concentrations primarily kill by clogging gill rakers and gill filaments (Bruton, 1985). Concentrations of suspended solids between 100-1000mg l^{-1} are harmful to the gills of fish, and the average concentration should not exceed 80 mg l^{-1} for the long-term health of fish (Alabaster & Lloyd, 1982; cited in Best, 1994).

(2) By reducing the suitability of spawning habitat and hindering the development of fish eggs, larvae and juveniles; all of these stages appear to be more susceptible to suspended solids than adult fish (Chapman, 1988; Moring, 1982). When fine sediment settles it can damage spawning areas by physically covering and 'cementing' gravel redds, trapping fry and reducing the oxygen supply to fish in their early life stages (Richards, 1985).

(3) By modifying the natural migration patterns of fish (Alabaster & Lloyd, 1982; cited in Wood & Armitage, 1997);

(4) By reducing the abundance of food available to fish due to a reduction in light penetration and as a result photosynthesis, primary production, and a reduction of habitat available for insectivore prey items (Bruton, 1985; Gray & Ward, 1982);

(5) By affecting the efficiency of hunting, particularly in the case of visual feeders (Bruton, 1985).

Organic pollution is the most common form of pollution in Ireland and it is the primary cause of most fish-kills in Ireland (McGarrigle *et al.*, 2002). Negative impacts on salmonids, such as decreases in abundance and condition of 0+ (young-of-the-year) fish and one year old fish, were found at those clearfell sites where negative impacts on macroinvertebrates also had been found, and were also predominantly associated with relatively large and longer-term increased levels of suspended solids and inorganic sediment (Johnson *et al.*, 2000).

d) Sediment and Margaritifera

Sedimentation can impact on any life stage of the freshwater pearl mussel *Margaritifera margaritifera* (FPM), but the first five years, during which the juvenile mussels are buried in the sand and gravel, are by far the most sensitive (Moorkens, 2000). Sediment clogs the gaps in the river bed, impeding the movement of water. This results in the death of juveniles by oxygen starvation. Plumes of sediment can also cause adults to close up in order to prevent clogging of their gills. Prolonged closure can lead to the death of the adults through oxygen deprivation or starvation. Less extreme conditions can also have adverse impacts, for example, by causing the female to prematurely release the glochidia (Moorkens, 2000).

3.3.3 Harvest area

Ideally, the more sensitive the site is, the smaller the portion of the catchment should be felled. However, the extent to which the felling area can be practically reduced is influenced by factors including the portion of the catchment planted and the age of the trees. The risk of windblow is a significant consideration as it also represents a threat in terms of sedimentation and nutrient loss. It is important that the benefits gained by a reduction in the proportion of catchment felled are not offset by increased sediment and nutrient losses arising from windblow.

The cumulative effect of all operations in the catchment must be considered, as individual impacts may be small but the combined impact may be significant. The felling programme in the catchment may comprise one or several coupes. As the difficulty of preventing sediment and nutrient losses tends to increase with increasing coupe size, smaller coupes are preferable. In Ireland, coupe sizes are generally distinguished by coupes under 25 ha (recommended for a general felling licence) and coupes over 25 ha (recommended for a limited felling licence (Forest Service, 2000b). Slope and runoff are important factors to consider when identifying coupe size. For example, large felling coupes (> 25 ha) may be acceptable on flat terrain or valley bottoms, whereas felling in very sensitive landscapes should be limited to 5-15 ha in accordance with Forest Service guidelines (Forest Service, 2000b).

During logging, the forest floor and mineral soil are disturbed or rutted to varying degrees depending on type of equipment used, slopes, layout of operations, intensity of harvest, soil type, weather, and other factors (e.g., Hornbeck *et al.*, 1986). Forwarding is the most common method of thinning/clearfell extraction used in Ireland (Forest Service, 2000b). Skidding is sometimes used for larger tree length material or in small-scale operations while cable extraction is confined to steep slopes or to very soft ground conditions (Forest Service, 2000b). In America the most common extraction method is skidding (NCASI, 2001) which has been shown to cause greater damage to freshwaters (Aust & Blinn, 2004).

In the US, accelerated soil erosion (the detachment and mobilization of mineral-soil across a landscape) commonly results from forest felling and transporting logs from the forest to the forest road. Soil response to logging depends on many factors including: length and steepness of slope, soil texture (sand, silt and clay), soil organic matter, aggregation, drainage (e.g., depth to water table), and plant cover (Aust & Blinn, 2004). Soil erosion, like soil exposure, varies widely across a logging site (especially for example, between log landings and haul tracks).

One of the most famous studies examining the effects of clearcutting and herbicide application in a northern hardwood system was done at the Hubbard Brook Experimental Forest, New Hampshire (Likens *et al.*, 1970). All vegetation growing on an entire watershed was felled and left in place. Vegetative regrowth was prevented by the repeated application of herbicides. No road building, skidding, or timber removal occurred in the watershed. Increases in stream water turbidity following treatment were negligible. The Hubbard Brook work provides the most direct evidence that felling of trees per se is not the primary cause of erosion. The many studies done prior to and after the Hubbard Brook de-vegetation experiment clearly document the overwhelming contribution of haulage tracks as primary sources of erosion and sedimentation.

An extensive study on clearfelling was established at 16 large sites in south-west Ireland (Gallagher *et al.*, 2000; Johnson *et al.*, 2000). Impacts on physico-chemical parameters were not found at all sites in this study, and where they were they were of short duration. Changes post-felling were mostly related to release of nitrates, suspended solids, inorganic sediment and soil inputs associated with poor management practices during felling and subsequent operations (Giller *et al.*, 2002). Although buffer strips could be effective at preventing sediment input, they alone did not prevent the input of sediment, suspended solids and soil into the stream (Johnson *et al.*, 2000). This points to the catchment specific nature of the interactions and the lack of generalised patterns. Further to these results, an index was developed to measure the impacts of harvesting on hydrochemistry and stream ecology that overcomes the confounding effects of natural variation in space and time (Johnson *et al.*, 2005).

Gallagher *et al.* (2000) showed increases in suspended solids at ten of the sixteen clearfell sites. Some increases were short-term and clearly centered around the duration of the clearfell operation. Where long-term increases were found, they were generally associated with flood events post-felling, whereby soil was washed from the clearfell area into the stream. Increases in sediment and soil on the streambed, which originated from the clearfell operation, were found at seven out of the sixteen sites. Again, most of the increases were short-term, but it is not known to what extent the sediment became trapped within the gravel of the streambed.

However, it was also clear from the study that the presence of a buffer strip alone did not always prevent an input of suspended solids and sediment into the stream, particularly if there was a single direct link between the clearfell area and the stream, such as a run-off channel, a bank collapse or a crossing point for machinery (Gallagher *et al.*, 2000).

At one clearfell site, mechanical removal of woody debris from the stream channel caused bank collapse and a substantial input of soil. At another clearfell site, a very large and relatively long-term input of suspended solids was recorded due to the overflow of a sediment trap. Effects of the sediment trap overflow were observed 2.4 km downstream of the clearfell area, and on one occasion sediment was observed to travel 4 to 5 km from the clearfell area (Gallagher *et al.*, 2000; Giller *et al.*, 2002; Johnson *et al.*, 2000).

Turbidity and sedimentation caused by logging are water quality concerns for forested watersheds. The effects of sediment on fish are primarily caused by reductions in stream bed gravel permeability, resulting in reduced embryo survival, habitat loss due to the filling of pools, and reduced food supply due to the impacts of sediments on lower trophic levels (Acornley & Sear, 1999; Argent & Flebbe, 1999; Soulsby *et al.*, 2001). Logging operations can degrade lakes and reservoirs by filling them with sediment transported by streams and by increasing turbidity. Decreases in salmonid populations have been linked to forestry-related sediment inputs in Nova Scotia and New Brunswick (Grant *et al.*, 1986; cited in Stafford *et al.*, 1996). After logging, Garman and Moring (1991) found higher levels of fine sediment in gravel pools of the East Branch of the Piscataquis River.

Soil disturbance is more related to type of logging operation (i.e. the machinery used) than to silvicultural system (Swank *et al.*, 1986).

4. MITIGATING MEASURES

Irish forest guidelines, in existence since 1991, were created to provide best management practices to minimise adverse impacts of forest operations on the environment, in particular during site preparation, harvesting, and road construction (Forest Service, 2000a,b,c,d, 2001, 2002, 2003; Ryan *et al.*, 2004).

There are four main mechanisms by which the amount of nutrients and sediment reaching the aquatic zone can be reduced: 1) minimisation of soil disturbance; 2) settlement via sediment traps; 3) filtration via buffer/riparian zones; and 4) brash mats.

4.1 Minimisation of soil disturbance

An important way to reduce the nutrient loading on aquatic systems from forest lands is to control the proportion of a catchment disturbed at any one time. This applies to both N and P loads, and to reducing the delivery of erodable solids to streams. Cummins & Farrell (2003) suggested as a possible ideal that, over larger catchments, the yield of suspended solids, or of dissolved nutrients, should be constant over the whole forest rotation. This could be achieved by felling and reforesting in each year approximately that proportion of the catchment which is equal to the area of the catchment divided by the number of years in the rotation (Cummins & Farrell, 2003). The Forest Service currently limits clearfelling to coupes no greater than 25ha in area (Forest Service, 2000b). The Forest Harvesting and the Environment Guidelines recommend phased felling in a way that minimises cumulative effects and ensures that succeeding rotations do not have the same even aged structure. In highly sensitive areas, the coupe size should be minimised (Forest Service, 2000c).

The Code of Best Forest Practice (Forest Service, 2000b) advises to carry out harvesting operations from April to October, as ground conditions tend to be drier during this period. The Code also suggests that mechanised operations during and immediately after heavy rainfall should be suspended on soils with a high erosion risk and/or low bearing capacity.

The transportation network (i.e., the soil impacts of vehicles and the layout of skid trails and roads) was the main factor that primarily determined soil exposure and impact from a study of 21 logging sites in North Carolina (Richter, 2000). Indeed, in the US, road construction and vehicle passes over streams have been widely shown to be the greatest threat to water quality in forests, mainly through sedimentation (Aust & Blinn, 2004).

The efficacy of the British Forestry Commission's Forests and Water Guidelines in controlling diffuse pollution from forestry was tested in two catchments in Argyll West Scotland (Nisbet *et al.* 2002). Both studies demonstrated that good forest management can effectively control the threat of diffuse pollution within sensitive catchments. At one site, the extensive ploughing and drainage of the catchment peaty soils caused little disturbance to the freshwater environment with no apparent effect on the macroinvertebrate fauna (Nisbet *et al.*, 2002). The key measures believed responsible for minimising the impacts of initial site preparation were "the shallow nature of the ploughing, which was successful in limiting the exposure of the more erodible, underlying mineral subsoil; the use of furrow-end buffer strips on steeper slopes (>5°), which were efficient at retaining any mobilised sediment; and the careful control of land drainage, which prevented the overloading and consequent erosion of the plough furrows and drains" (Nisbet *et al.*, 2002). It was also recognised that these features were often the cause of "dirty water" problems associated with past afforestation in the locality and elsewhere (Stretton, 1984; Richards, 1985). In Ireland, recent research examining the efficacy of the Forest Service guidelines has provided similar results. On a blanket peat catchment in Co. Wicklow with harvesting being undertaken, there was found to be no elevated phosphorus concentrations or transported phosphorus below the forest in comparison with conditions above the forest (Machava *et al.*, 2007). Additionally, the authors also investigated the impacts of afforestation on water quality on land that was previously used for agriculture (cambic podzol soil) in Crossmolina, Co. Mayo. Again, no adverse impacts of forest operations were detected. In fact, P concentrations in the surface water were actually lower after afforestation than before (Machava *et al.*, 2007). These studies suggest that if Forest Service guidelines are strictly adhered to then negative impacts on water quality should be neglible.

4.2 Sediment traps

Mobilised sediment transportation can be minimised by the use of naturally occurring vegetated overland flow areas and the use of sediment traps. Sediment traps should be installed at intervals, ideally as close as possible to the source of the sediment. Each site needs to be assessed regarding sediment management and appropriate plans adopted. Where required, correctly planned, installed and maintained sediment traps/drains for each individual site will help to ensure that water quality is protected (Ryan *et al.*, 2004).

4.3 Buffer/riparian zones

Reducing the detrimental effects of forestry upon rivers requires a buffer area, physically separating the watercourse from ploughing, planting and harvesting operations. Buffers that are too narrow are likely to provide inadequate protection, while if they are too wide the area of productive crop, and subsequent income, are reduced (Broadmeadow & Nisbet, 2004). Therefore a balance needs to be reached between the benefits and costs of increased riparian buffer widths, based on the considerations of the main functions of the riparian buffer in relation to the sensitivity of a given site (Broadmeadow & Nisbet, 2004).

The success of buffer zones at retaining P depends on the mechanisms by which P is transported from land to stream (Muscutt *et al.*, 1993). If the major P fraction is particulate, or at least surface runoff-derived, buffer zones need to trap and retain this sediment-associated P. P retention in riparian soils is influenced by soil organic matter, pH, and Fe and Al content, and these factors exhibit spatial variability across a landscape (Lyons *et al.*, 1998). In this case, factors such as the roughness coefficient of the vegetation in the buffer zone are critical. Vegetation characteristics will vary seasonally and it will be important to match maximum vegetation trapping efficiency with the main periods of P export (Muscutt *et al.*, 1993). However, P export in surface run-off will be primarily associated with periods of high rainfall, commonly in winter months. This coincides with periods where vegetation cover and trapping efficiency may be least. If P is transported in subsurface flows, where macropores or subsurface systems exist, then certain buffer zones will be ineffective at retaining P. It has also been suggested that as time progresses the soils in the buffer strip may become saturated with P and no longer retain it effectively but, at the moment, this appears to be a prediction based on the behaviour of some soils rather than an observed phenomenon (Hilton, 2003).

Some riparian zones have the ability to retain large amounts of N. The ability of a riparian zone to remove large quantities of nitrate from water or soil is influenced by the type and quantity of existing vegetation and by soil conditions (Jordan *et al.*, 1998). The microbial flora and fauna have an important part to play in nutrient retention, especially in regulating denitrification (Martin *et al.*, 1999). Riparian zones contain potential hotspots of denitrification due to the presence of high water tables that may produce the anoxic conditions required for the process to occur (Hill, 1996). Lowrance *et al.* (1984) observed a 31.5 kg/ha/yr loss of nitrogen from the riparian zone, reducing the N export by the watershed to 13.0 kg/ha/yr. This is a key removal pathway for nitrate and tends to be greatest during winter when soils are wettest and vegetation uptake is at a minimum. The process is aided by the availability of easily mineralisable sources of organic carbon from leaf litter, root decay and root exudates (Broadmeadow & Nisbet, 2004).

Many factors influence the ability of buffer zones to remove sediments from land runoff, including the sediment size and loads, slope, type and density of riparian vegetation, presence or absence of a surface litter layer, soil structure, subsurface drainage patterns, and frequency and force of storm events (Osborne & Kovacic, 1993). Riparian buffers must be properly constructed and regularly monitored in order to maintain their effectiveness. Probably the most important consideration is the maintenance of shallow sheet flow into and across the buffer. Where concentrated flow paths begin to form or deep sediments begin to accumulate, the buffer can no longer maintain its filtering ability. In southern Scotland, buffers of coarse grass and heather were effective at reducing run-off flow velocities and increasing the deposition of suspended solids released after pre-planting ploughing and drainage operations (Swift & Norton, 1993). It was estimated that 50% attenuation in the suspended sediment load was achieved across a buffer 60-70m wide on mineral soils, when the vegetation was growing actively.

Nieminen *et al.* (2005) tested the efficiency of riparian buffer zone areas to reduce the concentrations of suspended solids in discharge from peatlands drained for forestry in south-central Finland. The two largest buffer zones (1.01 ha and 1.03 ha) reduced the concentrations of suspended solids by > 70%. The efficiency of three medium-sized buffer zones (0.20 ha, 0.16 ha and 0.12 ha) was 50–60%, but no reduction occurred at the smallest two buffer areas (0.09 ha) (Nieminen *et al.*, 2005). The use of buffer zones in reducing sediment load from peatlands drained for forestry purposes is recommended, but relatively large areas for efficient removal capacity would be needed. Wenger (1999) provided a review of the capability of riparian buffers to confine suspended solids, in studies undertaken in the US. The broad concensus was that riparian buffers ability to trap suspended solids was positively correlated with width and negatively correlated with slope (Table 4.).

Vegetation structure has a direct effect on the level of shade cast over the water's surface and thus on water temperature. This is of great importance as water temperature is fundamental to the physiology and behaviour of fish (Solbé, 1997). Guidelines seek to achieve a balance between excessive and no shade of riparian vegetation. For example, the Forest Service 'Forestry and Water Quality Guidelines' recommend that 50% of the water surface should be under dappled shade (Forest Service, 2000a). Trees which cast heavy shade, including native species, should be planted sparingly, if at all (Dobson & Cariss, 1999). Different aquatic taxa have different requirements, and many require the soft margin habitats associated with unshaded river stretches (Weatherley *et al.*, 1993).

Table 4. Riparian buffer width, slope and total suspended solids (TSS) removal rates. The ability of riparian buffers to trap suspended solids is positively correlated with width and negatively correlated with slope. Adapted from Wenger (1999) and references therein.

Author	Width (m)	Slope (%)	% Removal of TSS
Dillaha et al. (1988)	4.6	11	87
Dillaha et al. (1988)	4.6	16	76
Dillaha et al. (1988)	9.1	11	95
Dillaha et al. (1988)	9.1	16	88
Dillaha et al. (1989)	4.6	11	86
Dillaha et al. (1989)	4.6	16	53
Dillaha et al. (1989)	9.1	11	98
Dillaha et al. (1989)	9.1	16	70
Magette et al. (1989)	4.6	3.5	66
Magette et al. (1989)	9.1	3.5	82
Peterjohn & Correll (1984)	19	5	90
Peterjohn & Correll (1984)	60	5	94
Young et al. (1980)	21.3	4	75-81
Young et al. (1980)	27.4	4	66-93

A number of riparian transect studies have documented substantial nutrient filtering along hydrologic flow paths (Peterjohn & Correll, 1984; Lowrance *et al.*, 1985; Jacobs & Gilliam, 1985; Cooper, 1990; Jordan *et al.*, 1993), yet other studies have described more variable results (Osborne & Kovacic, 1993; Altman & Parizek, 1995; Hill, 1996; Correll *et al.*, 1997, cited in Baker *et al.*, 2006; Vidon & Hill, 2004). In whole watershed analyses, evaluations of riparian effects have been mixed, showing strong (Weller *et al.*, 1996; Johnson *et al.*, 1997; Norton & Fisher, 2000; Jones *et al.*, 2001) or weak (Omernik et al., 1981; Osborne & Wiley, 1988) riparian effects. Quantifying or even just demonstrating a riparian effect on nutrient discharges across a river network, among watersheds, or across regional landscapes is still a challenge.

In a study to determine the effect of ground cover on sediment and P export from pastured riparian areas under simulated rainfall events, Butler *et al.* (2006) demonstrated that riparian bare areas could contribute substantial sediment (>215 kg ha^{-1}) and P (0.7 kg P ha^{-1}) to surface waters during heavy rainfall, whereas export may be reduced equally well by low cover (45%) as by high cover (95%) of ground vegetation.

4.4 Brash mats

Forest Service guidelines recommend that dense brash mats should be created and maintained on all machine routes, to avoid soil damage, erosion and sedimentation (Forest Service, 2000b,c). Most damage is caused by forwarding operations on soils which are naturally poorly drained. These soils include surface-water gleys, shallow peats over-lying poorly drained clay soil, and deep peats. It has been recommended that brash mats should also be renewed before they become too worn as sites with insufficient brash are unsuitable, while poorly managed extraction routes will result in recovery of low quality brash material and a high risk of rutting (Hutchings *et al.*, 2002).

Recent research has demonstrated that properly constructed and maintained brash mats formed from conifer harvesting residues are highly effective in soil protection (Hutchings *et al.*, 2002; Wood *et al.*, 2003). Hutchings *et al.* (2002) studied various thicknesses of brash mat and their ability to reduce compaction of a surface water gley soil at Kielder Forest, Northumberland. The authors found that the soil under brash mats experienced some compaction to at least 45cm depth. While the thickest brash mat, composed of residues from 10 rows of trees, was unable to prevent compaction completely, its protective role was obvious when compared to timber extraction over bare soil, where penetration resistance was encountered at 6cm depth (Hutchings *et al.*, 2002). Soil can be protected by the use of brash mats along routes where machinery is obliged to travel although even thick brash mats were unable to prevent compaction completely (Hutchings *et al.*, 2002). What remains unclear is at what point does compaction have a detrimental effect on the establishment and stability of future tree rotations.

5. KNOWLEDGE GAPS

We have an understanding of the mechanisms of eutrophication and sedimentation but it is apparent that there is insufficient information on the realised impacts on freshwaters. As a result, while streams in Irish forests have the potential to experience some degree of nutrient and sediment inputs, particularly on upland environmentally sensitive areas, the percentage of these that are impacted, and the extent and duration of same, remains unknown.

5.1 PRESSURE

An important question to ask is what proportion of phosphorus actually reaches freshwater systems within a managed forested catchment and what percentage of P is retained in the soil and vegetation? Mineral soils bind P, but what is unclear is how much mineral soil in a catchment is sufficient to take up P. This leads us to ask, do these soils become P saturated? This is of obvious concern as soils saturated in P will potentially release excess P into watercourses. Therefore, the relationship between P loss and percentage mineral soil needs to be quantified. Models need to be developed for nutrient and sediment losses under different soil, slope, climatic conditions etc. While the efficacy of mitigation measures (e.g. buffer zones) have been assessed both in Ireland and abroad it is still unclear what physical and biological factors characterise an effective buffer zone. It has also been suggested that as time progresses the soils in the buffer strip may become saturated with P and no longer retain it effectively but, at the moment, this appears to be a prediction based on the behaviour of some soils rather than an observed phenomenon (Hilton, 2003). The question raised therefore is, are the recommended buffer widths and riparian vegetation effective at mitigating impacts of eutrophication and sedimentation in Irish forests?

5.2 RECEPTOR

It is possible to get an initial indication of areas likely to be sensitive but whether or not an adverse impact will occur in these areas is far from certain. Studies are required to establish if eutrophication is a widespread problem in upland streams and to what extent it occurs. We need to address if detected nutrient loadings have a biological impact and what is the relative magnitude of this impact for stream communities and salmonid populations? Increased nutrients could potentially be beneficial for salmonids but detrimental to other organisms showing that not all species will react to nutrients in the same manner. The temporal and spatial scales of the impacts (short-term or long-term, short-distance or long-distance) are also not known (i.e. how far downstream does any impact last in terms of time and distance). Another question that needs clarifying is which freshwater system exhibits the greatest sensitivity to nutrient loadings (e.g. are streams less vulnerable than lakes)? Within streams, factors that potentially mitigate against P include slope and shade, but it is unknown to what extent this occurs.

There is also a knowledge gap involving the macroinvertebrate community response to the twin pressures of eutrophication and acidification. Do they respond synergistically or antagonistically (i.e. is it an additive response or not)? Finally, as acquiring a bio-indicator to detect stream disturbance is of great importance we must ask what is the most appropriate bio-indicator group for detecting impacts in small upland streams? For this reason, are macroinvertebrates the most appropriate bio-indicator group for detecting eutrophication impacts utilising the Q value system, as this system is generally evaluated on third order streams and higher, whereas upland forest streams will be of smaller first and second order. Diatoms have been suggested as a more appropriate bio-indicator group as they have been found to be more sensitive to eutrophication (Kelly & Whitton, 1995).

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