Forestry Commission

Research Note

Forestry and surface water acidification

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Forests and forest management practices can affect surface water acidification in a number of ways. The primary mechanism is the ability of tree canopies to capture more sulphur and nitrogen pollutants from the atmosphere than other types of vegetation. Pollutant scavenging is expected to have peaked in the 1970s when emissions were greatest and led to surface waters draining catchments dominated by forestry being more acidic. The introduction of emission control policies in the 1980s has achieved major improvements in air quality and studies show forest sites to be recovering in line with their moorland counterparts. However, forest streams remain more impacted, requiring continued restrictions on new tree planting and restocking. Tree planting can influence acidification by the scavenging of acid deposition, base cation uptake, the scavenging and concentration of sea salts, soil drying and the formation of an acid litter layer at the soil surface. Cultivation, drainage and road building, fertiliser use, felling and harvesting, and restocking also have effects. This Research Note considers each of these factors in turn and assesses the role of tree species, planting scale and design. It covers the identification and protection of vulnerable areas, use of critical load and site impact assessments, research and monitoring, and measures to promote recovery. Continued monitoring will be essential to demonstrate whether current measures remain fit for purpose and guide the development of good practice.

Forest Research

Introduction

The role of forestry in acidification of freshwaters has been a prominent issue in the UK since the late 1970s. Concern that conifer forests could be contributing to the 'acid waters problem' that affected large parts of the UK uplands arose following a number of catchment studies in Scotland and Wales in the early 1980s (Nisbet, 1990). These found evidence of higher levels of acidity or dissolved aluminium in waters draining afforested compared to moorland catchments (Harriman and Morrison, 1982; Stoner and Gee, 1985). The timing of the acidification and decline of fish populations after a period of major upland conifer afforestation in the 1960s and 1970s was also cited as support for a forestry effect.

The relative contribution of air pollution verses forestry to acidification was hotly debated during the 1980s. By dating changes in fossil diatom populations contained within stratified lake sediments, Battarbee *et al*. (1988) were able to show that acid deposition was the principal cause of the problem. Acidification usually commenced before the period of upland afforestation and was largely absent in sensitive forested catchments in areas receiving low levels of acid deposition, such as northwest Scotland (Battarbee, 1988). However, some diatom studies found evidence of an acceleration of acidification following conifer afforestation, suggesting that forestry could be a contributory factor.

The Department of the Environment and the Forestry Commission held an expert workshop in 1990 to examine the interaction between forestry and surface water acidification. Of a range of potential causal mechanisms, the ability of forest canopies to enhance the capture of acidic pollutants from the atmosphere was considered to be the most important (Department of the Environment, 1991). The workshop report recommended that the Forestry Commission should include this pollutant 'scavenging' effect when assessing conifer afforestation schemes and should adopt the critical loads approach to identify waters at risk. The report also highlighted a need for additional long-term studies to quantify the forest effect and the response of waters to planned emission reductions.

The Forestry Commission amended its 1988 *Forests & water guidelines* in 1991 to incorporate the critical loads approach (see below), with further revisions in 1993, 2003 and 2011 to ensure that the guidelines continued to reflect the results of recent research and experience. Catchment-based critical loads assessments were introduced for new planting in 1993 and were extended to include restocking proposals in 2003. Developments in mapping have focused on better targeting of waters at risk, leading to the recent shift away from indicative critical load maps to prioritise 'water bodies' that are failing or at risk of failing Good Ecological Status due to acidification caused by acid deposition.

This Research Note sets out the science underpinning guidance on managing forestry in acid-sensitive water catchments by describing how forests and forestry management practices can affect surface water acidification. The effects of new planting, cultivation, drainage and road building, fertiliser use, felling and harvesting, and restocking are considered in turn. Sections also cover the identification of vulnerable areas, use of the critical loads approach, research and monitoring, and measures to promote recovery.

Interaction between forestry and surface water acidification

New planting

The planting of trees can influence acidification in a number of ways, including by the scavenging of acid deposition, base cation uptake, the scavenging and concentration of sea salts, soil drying and the formation of an acid litter layer at the soil surface (Department of the Environment, 1991). Each of these is considered below, followed by an assessment of the role of tree type and species, and planting scale and design.

Pollutant scavenging

The primary mechanism responsible for a forestry acidification effect is the ability of forest canopies to capture more sulphur and nitrogen pollutants from the atmosphere than shorter types of vegetation (Department of the Environment, 1991). This increased capture, often termed scavenging, is primarily a function of stand height/structure, which creates turbulent air mixing and increases the rate at which pollutants are deposited onto forest surfaces (Fowler *et al*., 1989). The effect therefore becomes more important as trees grow and the height of the stand increases.

Trees can enhance the capture of atmospheric sulphur and nitrogen pollutants in gaseous (e.g. sulphur dioxide, nitrogen dioxide, ammonia and nitric acid), particulate and cloud water forms (Figure 1; Nisbet *et al*., 1995). Gaseous and particulate forms are referred to as 'dry deposition' and are usually greatest downwind of emission sources. Gases can either be absorbed via leaf stomata or deposit directly by reacting with leaf surfaces. Particulate deposition is generally slow due to the small size of atmospheric particles and thus relatively unimportant in the scavenging process (UKRGAR, 1990).

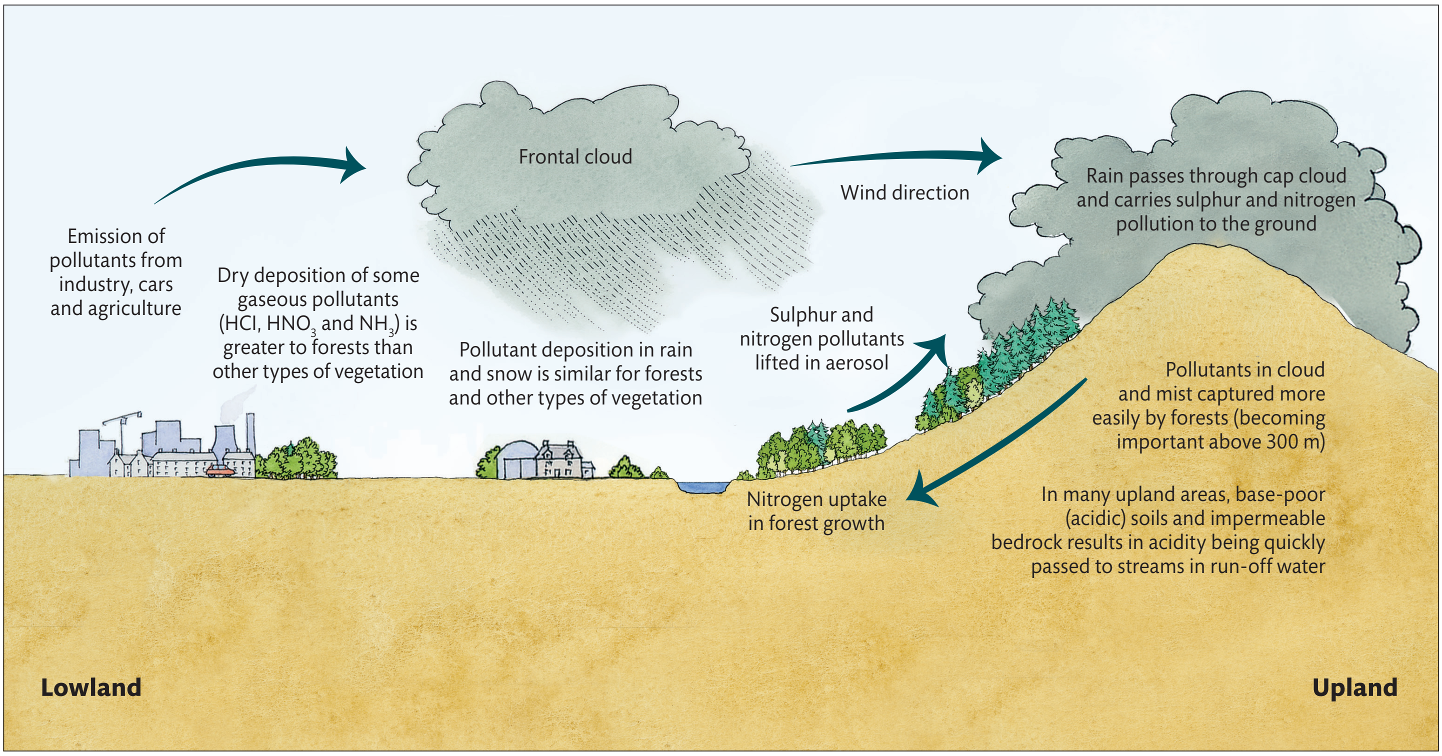
Cloud water or 'occult' deposition results from the reaction of pollutant gases with cloud droplets, which can generate high pollutant concentrations. The importance of this pathway increases with altitude and cloud duration, making it a key factor in upland areas. An analysis of cloud immersion frequencies across the UK found that cloud duration increased from 2–6% at 150 m altitude to 15–25% at 600 m (Weston, 1992). Modelling using cloud base observations suggests that occult deposition can become an important contributor to total pollutant deposition on land above 300–400 m elevation (UKRGAR, 1990). The 300 m contour was selected in the second edition of the *Forests & water guidelines* (Forestry Commission, 1991) as a lower limit for considering the impact of conifer afforestation but was removed from the fourth edition (Forestry Commission, 2003) due to concern that lower altitude forests could still exert a significant effect on sensitive waters.

Pollutant scavenging by young trees is minimal and only becomes significant after they begin to close their canopies. This is supported by catchment studies, which have found little evidence of a forestry acidification effect in 'young' (<15 years old) forested catchments (Stevens *et al*., 1994; Stevens *et al*., 1997). The timing of canopy closure varies with tree species, site type and altitude, but generally occurs around 15 years of age for conifers in the uplands. Acid water surveys have found the positive relationships between forest cover and different indices of stream acidity to be dominated by catchments with older (>30 years) conifer forests and in the case of nitrogen, those aged >40 years (Stevens *et al*., 1994; Stevens *et al*., 1997).

The level of pollutant concentrations in the atmosphere either as gases or in cloud water has a direct bearing on the importance of the scavenging effect. As the level of pollutants reduces, the scavenging effect becomes smaller in absolute terms. This explains why forestry has been found to have a minimal effect on surface water acidification in acid-sensitive areas receiving low pollutant deposition, such as in northwest Scotland (Kernan *et al*., 2010). It is also a key factor concerning the response of acidified forested catchments to emission control; as pollutant levels decline, so too will the significance of the scavenging effect and the contribution this makes to critical load exceedance (see below).

The efficiency of the forest scavenging effect varies between pollutants and is greater for nitrogen compared to sulphur due to the more reactive nature of nitrogen gases, as well as the tendency for a higher proportion of nitrogen pollution to be in gaseous ('dry') rather than dissolved ('wet') forms compared to sulphur. Modelling studies provide typical enhancement factors for pollutant capture by closed canopy forest stands of between 20 and 30% for sulphur and 50 and 100% for nitrogen (UKRGAR, 1990). Based on the most recently available modelled deposition data (UK 5 km grid data for 2006–08 from the Centre for Ecology and Hydrology), conifer forests increase the deposition of oxides of nitrogen (NO_X) by an average of 106% relative to moorland and ammonia (NH_X) by 58%, giving an average overall nitrogen deposition enhancement of 74%. The average enhancement for non-marine sulphur deposition is 22%. However, these figures hide considerable spatial variation around the means, with highest values generally observed in lowland areas closest to pollutant sources.

Figure 1 Interactions between forests and acid deposition.



Base cation uptake

A second way that tree planting can exacerbate acidification is through the uptake of base cations (calcium, magnesium, sodium and potassium) from the soil. Trees require base cations for growth and, if their removal exceeds the rate of replenishment from soil mineral weathering, inputs of leaf litter and dead wood, and deposition from the atmosphere, this will reduce the buffering capacity of the soil. A lower buffering capacity means less base cations to exchange with and neutralise acid deposition, potentially resulting in lower pH and/or higher aluminium concentrations in drainage waters. Base-poor, slow weathering soils are most vulnerable to this effect. Base cation uptake tends to be greatest during the first 15-20 years of tree growth as the young forest is building its leaf canopy. Once trees reach the canopy closure stage their base cation needs are to a large degree met through canopy recycling (Miller and Miller, 1987).

There is evidence to suggest that trees could partly offset the acidifying effect of base cation uptake by the action of tree roots enhancing soil mineral weathering, especially at depth through deeper rooting (Vejre and Hoppe, 1998). An increased release and mobilisation of base cations would help to buffer acid inputs, although the supply of base cations will be constrained within shallow upland soils. However, deeper rooting could also potentially exacerbate the situation by exposing and mobilising aluminium sources, leading to higher aluminium concentrations in drainage waters.

Sea salts

As well as scavenging air pollutants, tree canopies can be effective at enhancing the deposition of sea-salt aerosols from the atmosphere. This effect is greatest along west-facing, upland, coastal areas exposed to Atlantic storms and is thought to be largely responsible for the observed greater concentrations and export of sodium and chloride in stream waters draining forested compared with moorland catchments (Stevens *et al*., 1997). Another factor contributing to the higher sea-salt concentrations is the increased evaporation of water from forests, especially conifer, due to the greater water use by trees (Nisbet, 2005).

While sea salts are neutral, they can influence water acidity through chemical interactions with the soil, especially involving cation exchange sites. Large inputs of marine cations such as sodium and magnesium can displace hydrogen and aluminium from exchange sites in acidified soils, leading to a transient increase in the acidity of drainage waters but not long-term acidification (Evans *et al*., 2001). Such sea-salt-driven acid events have been associated with fish kills in acidified catchments (Teien *et al*., 2004).

It has been argued that the enhanced deposition of sea salts to forests could increase the severity of acid episodes and explain why forest stream waters can be more acid than their moorland counterparts in some lower acid deposition areas (Johnson *et al*., 2008). However, there is limited evidence to support such an effect. Soils tend to become saturated with sea salts during sea-salt-driven acid events and it is questionable whether the forest enhancement factor could exert a significant additional effect. In addition, frequent Atlantic storms condition soils with sea salts making acid exchange less likely, while there is evidence of waters in affected areas showing increasing signs of recovery, suggesting that acid deposition is the main cause of acid impacts.

Soil drying

Summer drying of organic soils can lead to increased oxidation of soil organic matter and the release of organic acids and stored sulphate. The washout of these chemicals following heavy rainfall has the potential to generate acid events in stream waters. Since forest drainage and the higher water use by trees can enhance soil drying this could contribute to greater acidification. While there is some evidence that the episodic washout of stored sulphate after droughts can be biologically damaging (Evans *et al*, 2014), studies suggest that the additional acidifying effect associated with forest activities is relatively small and unimportant (Department of the Environment, 1991).

Litter horizon

Forest soils are typically acidic unless underlain by calcareous rock. They are often characterised by having acid surface litter, fermentation and humus layers, due to the enrichment of the soil with organic matter (Forestry Commission, 2011b). The release of organic acids from these layers can reduce the pH of surface drainage waters but this natural organic acidity reflects normal soil processes and rarely leads to adverse effects on receiving stream waters (Department of the Environment, 1991).

Tree type and species

Although little studied, differences in tree type and species appear to have a relatively small influence on the primary mechanism responsible for a forest acidification effect. This is because pollutant scavenging is driven by the aerodynamic roughness of the tree canopy, which mainly reflects canopy height, structure and to a lesser degree, leaf area index. No distinction is made between conifer or deciduous trees in deposition model applications due to the assumed similarity in canopy roughness. This is supported by throughfall studies showing little difference in acid deposition inputs between adjacent conifer spruce and deciduous larch stands (Reynolds *et al*., 1989), although is less likely to hold where site and management factors lead to marked differences in forest canopy height and form. For example, Gagkas *et al*. (2011) calculated a lower canopy roughness length (0.73 m) for an upland, open, birch woodland at Loch Katrine in Scotland than that usually applied to conifer forest (1.0 m). Nevertheless, the birch value is still significantly higher than those associated with moorland vegetation (0.1–0.2 m; UKRGAR, 1990).

Forest type and species can also affect the other acidification processes, the most significant of which is base cation uptake. The impact on the latter mainly relates to differences in growth rates and associated changes in wood density and base cation concentrations in biomass. In general, the faster growth rate of conifers is offset by the correspondingly lower wood density and base cation concentrations compared to broadleaves, but there is significant variation between species. For example, while measurements at the UK's long-term forest monitoring Level II sites show annual average base cation uptake to be higher for broadleaves on equivalent base-poor soils (0.32 keq ha^{-1} yr^{-1}, compared to 0.27 keq ha^{-1} yr^{-1} under conifers), values varied greatly between species and reached a high of approximately 0.40 keq ha^{-1} yr^{-1} for faster growing Sitka spruce (Langan *et al*., 2004).

Related factors include evidence of deeper rooting by broadleaves promoting the cycling of base cations in the soil (Collier and Farrell, 2007), although this is likely to be less important in shallow upland soils. Another factor is the influence of the previous widespread use of potash and calcium phosphate fertiliser to aid the establishment of Sitka spruce plantations on nutrient-deficient soils, which will help to compensate for their potentially higher base cation uptake rates.

Forest type can also influence acidification through the contrasting effects of conifers and broadleaves on soil nitrogen processes. Recent studies suggest that, as well as maintaining higher nitrogen uptake rates, broadleaved woodland soils are better able to retain nitrogen due to differences in the cycling of soil organic matter beneath the two tree types (Tipping *et al*., 2012). This makes broadleaved woodland soils less likely to become nitrogen saturated and contribute to acidification through leaking nitrate.

Lastly, the planting of nitrogen-fixing species such as alder can promote nitrate leaching from soils and contribute to surface water acidification. Gagkas *et al*. (2008) showed that nitrate leaching from alder woodland was associated with significantly higher aluminium concentrations in stream waters. It is for this reason that the UKFS Guidelines on *Forests and water* (Forestry Commission, 2011a) recommend limiting the planting of alder to less than 10% of the area within riparian zones in vulnerable catchments.

Planting scale and design

The scale of forest cover within a catchment has a direct influence on the magnitude of the acidification effect. Studies in upland Wales in the 1980s and 1990s found that it was difficult to detect any impact on streamwater chemistry in catchments with less than 30% cover of closed canopy conifer forest, but above this threshold there appeared to be a direct relationship between proportion of forest cover and the level of stream water acidity (pH and/or aluminium) (Ormerod *et al*., 1989; Stevens *et al*., 1997). Similar results have been found in regional studies in Scotland (Dunford *et al*., 2012). Although catchment studies have been unable to examine the effect of altitude on this relationship, it is likely that higher altitude stands will have a disproportionately greater effect for a given area of forest cover.

Planting design can exert a significant influence on the contribution of forestry to acidification through forest type and species, as well as the amount and distribution of open space, and the extent of forest edge. Targeting open space to higher altitude parts of a site, where pollutant scavenging is most enhanced, could be particularly beneficial. The same has been argued for riparian zones due to the proximity of scavenged pollutants to the watercourse, although this has not been borne out by riparian clearance studies. However, planting riparian zones with an open canopy of native broadleaved woodland, in accordance with the UKFS Guidelines on *Forests and water* (Forestry Commission, 2011a), has been shown to benefit riparian and aquatic habitats and aid biological recovery of acidified waters (Broadmeadow and Nisbet, 2004).

The size and shape of individual forest blocks directly affects atmospheric deposition by dictating the extent of forest edge, which increases aerodynamic roughness and canopy scavenging. Neal *et al*. (1994) found sea-salt capture to be greatly enhanced at and within 20–50 m of the forest edge, especially along windward-facing sides. Somewhat surprisingly, the edge effect was small for the capture of sulphur and nitrogen pollutants but it has been shown by others to be substantial in locations close to pollutant sources, such as immediately downwind of ammonia emissions from intensive livestock-rearing units (Sutton *et al*., 2004). Edge effects become increasingly 'diluted' as the size of individual forest blocks increases and are likely to become marginal on an overall area basis for blocks larger than a few hectares in extent (Neal *et al*., 1991).

Cultivation, drainage and roads

Soil cultivation and drainage can influence acidification in a number of positive and negative ways. Cultivation disturbs and mixes the soil, potentially improving soil aeration and warming the soil. This can promote the weathering of minerals, especially derived from deeper within the soil, leading to increased buffering of acid deposition. However, it can also increase acidity by enhancing the oxidation of organic matter and sulphides, generating increased leaching of sulphate, nitrate, aluminium and/or dissolved organic carbon.

Another potentially significant effect of cultivation is to alter soil water pathways and in particular to direct more water to depth within the soil. This can be most marked where cultivation disrupts a compact or sealed layer such as an ironpan, allowing water to access and drain through relatively more base-rich, mineral subsoil horizons compared to previously flowing laterally through acidic, organic, upper soil horizons. While a shift to slower, deeper water pathways will generally result in better buffering of drainage waters, it can have the opposite effect where mineral subsoils are very acidic and provide a source of aluminium. This factor could partly explain the higher aluminium concentrations found in some soil and stream waters draining conifer plantations. For example, Reynolds *et al*. (1988) found aluminium concentrations to be 1.5 to 3 times higher in soil waters beneath conifer forest compared to those below grassland on the same stagnopodzol soil type, whereas hydrogen ion concentrations were more similar. Flowpath changes could have brought about a shift in the 'Gibbsite' equilibrium that occurs in soil solutions between hydrogen ion (i.e. acidity) and aluminium. This has also been observed in surface waters draining forest and moorland catchments in the same regions (Evans *et al*., 2014). Although the shift effectively reduces the effects of sulphate and nitrate leaching from forests on acidity, the corresponding increase in aluminium leaching could have deleterious effects on freshwater biota, including fish.

Forest drainage operations also change soil water pathways and potentially surface water acidity. As with cultivation, this can be either positive or negative. The exposure of more base-rich mineral layers can increase buffering of drain flows, as can the seepage of waters from deeper within the soil (Ramberg, 1981). Drainage operations are known to produce a long-term increase in the contribution of base/low flows from upland soils, which can initially be as much as 100% for large-scale drainage treatments (Robinson *et al*., 2003). However, forest drainage can also increase the speed of run-off by providing a more direct route for water to flow to streams. Studies show that this can shorten the time for flows to peak following a rainstorm by a third and increase peak height by 15-20%, making streams more responsive to heavy rainfall (Robinson, 1986). Since stream acidity usually peaks during high flows, reflecting the dominance of surface water pathways through more acid soil layers, it is argued that drainage has an overall negative impact on acidification (Langan and Hirst, 2004). Catchment-scale studies though have struggled to find evidence of drainage exerting a significant impact on stream acidity, probably due to the relatively small size of the effect and the diminishing contribution of drainage treatments as catchment size increases. It has also been shown that the effect of drainage on peak flows decreases with increasing peak size. Moreover, in time drains naturally infill due to soil subsidence and vegetation growth, and effects are difficult to detect after 20 years (Robinson *et al*., 2003).

There is little evidence of forest roads having a significant effect on surface water acidification at the catchment scale, although local impacts are possible. Much depends on the nature of the stone used to construct roads, with scope for the use of very acid or alkaline materials increasing or decreasing the acidity of road drainage waters, respectively. Another way that road construction can exert an impact is through the local quarrying of road stone. Changes to flowpaths or the exposure and subsequent oxidation of sulphide-rich deposits could generate highly acidic, metal-rich drainage waters that have the potential to pollute local streams.

All of the above potential impacts have been largely addressed by changes to forest practice linked to the development of the UKFS Guidelines on *Forests and water* (Forestry Commission, 2011a). Key measures include: the shift away from deep ploughing to more superficial forms of soil cultivation; better matching of tree species to site conditions, reducing the need for intensive drainage; leaving undisturbed buffer areas along watercourses; and separating road drainage from natural watercourses.

Fertiliser applications

Fertiliser use can affect acidification through the addition of base materials to the soil or by influencing nitrogen processes. Although little used in UK forestry, applications of nitrogen fertiliser in the form of urea or ammonia nitrate are potentially acidifying due to nitrification of ammonia and leaching of nitrate. Rock phosphate and potash were more commonly applied to nutrient-poor upland soils in the past to aid forest establishment and have a relatively neutral (potash) or neutralising (rock phosphate) effect on the soil. Rock phosphate contains 32–48% calcium as apatite (calcium phosphate mineral) plus free carbonates such as calcite and dolomite. These exert a liming effect, equivalent to 100–200 kg of lime per hectare, which can raise soil pH and base saturation by a small degree (<1.0 pH unit) (FAO, 2004). The use of these fertilisers is uncommon in second rotation forests.

Forest felling and harvesting

The impacts of forest felling and harvesting on acidification are many and complex. Felling brings a temporary halt to forest scavenging and thus a reduction in the capture of atmospheric pollutants and sea salts. There is also a cessation of base cation and nutrient uptake and cycling by the trees, increasing the availability of both for leaching from the soil. Tree removal results in a marked reduction in water use with the switch to a bare site, raising the soil water table and promoting surface run-off pathways and soil leaching. Added to this are the large inputs of needle/leaf and branch material in the form of felling brash, and the release of nutrients and soluble carbohydrates from these. Lastly, soil disturbance caused by forest harvesting machinery can influence soil processes such as weathering and leaching, as well as drainage water pathways. All of these effects vary depending on the scale and nature of felling and harvesting practices, the extent of past thinning, forest type and species, forest age, site and soil type, rapidity of revegetation and/or replanting, and the pollution climate.

A study by Neal and Reynolds (1998) of 51 clearfelling sites from across upland Wales found that, in the vast majority of cases, these various acidifying and neutralising effects tended to balance out at the catchment level, making it very difficult to discern any significant impact on stream water acidity when set against the background variation in water chemistry. The main exception concerned a proportion of sites (15–20%) where felling led to marked nitrate leaching, which lasted for a period of 2–5 years, depending upon the rate of revegetation. This is caused by the disruption to nitrogen cycling and resulting increased rates of mineralisation and nitrification in the soil. Sites dominated by brown earth soils appeared to be most associated with such a nitrate response but it also occurred across other acid soil types. While the increase in nitrate concentrations in soil and stream waters poses a negligible risk of exceeding drinking water standards, of more concern is the resulting decrease in acid neutralising capacity (ANC) and potential reduction in pH and/or increase in aluminium.

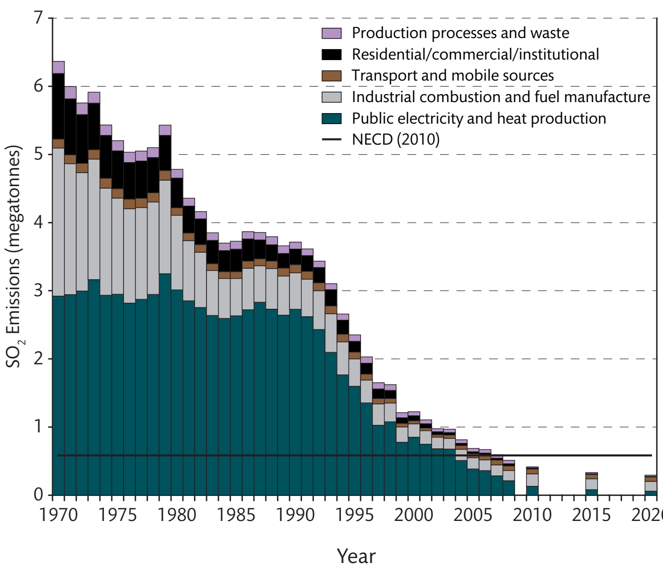
The observed decline in ANC following felling is usually less than 30 μeq l^{-1}, even when entire catchments are felled (Neal and Reynolds, 1998). This makes the acidification effect marginal and difficult to detect when 20% or less of a catchment is felled within a given year. As a result, the UKFS Guidelines on *Forests and water* (Forestry Commission, 2011a) selected 20% as a precautionary threshold for the extent of felling in order to protect potentially vulnerable sites from a nitrate-induced acidification effect. Current guidance recommends against felling more than 20% of an acidified catchment in any 3-year period unless a detailed site impact assessment shows stream waters to be protected from acidification. Adopting forest practices to speed up the revegetation of a felled site can help to reduce nitrate leaching, as can avoiding those that can have the opposite effect, such as filling spoil trenches with fresh brash or mulching brash.

The other significant issue concerning forest harvesting is the long-term risk of consecutive harvesting cycles contributing to soil and water acidification by the accumulated removal of base cations in harvested produce. Much depends on the overall balance between base cation inputs from the atmosphere and soil weathering, and losses in biomass and soil leaching. Losses in biomass generally represent a small contribution to the effect compared to the other mechanisms, making it difficult to determine the long-term outcome when set against the errors involved (Helliwell *et al*., 2011). Modelling suggests that it could take many rotations/several centuries for such an effect to become significant on most soils (Helliwell *et al*., 2011), with little evidence of observable impacts on soil base cations after the first forest rotation (Neal and Reynolds, 1998). The greatest risks are associated with very base-poor soils such as peats and the much higher base cation losses resulting from the removal of needles/leaves, branches and tree-tops in whole-tree harvesting or by harvesting tree stumps. For this reason, the UKFS Guidelines on *Forests and water* stipulate against these practices on high-risk soils (Forestry Commission, 2011a,b).

Restocking

Restocking will eventually lead to the same acidification mechanisms associated with the first rotation being re-established as the replanted crop develops. However, a major difference is that the second rotation will be exposed to a greatly improved pollution climate (Figure 2). This is highly significant for the primary scavenging mechanism since the enhanced capture of acid pollutants by the re-established forest canopy will be markedly lower in absolute terms. Many first rotation upland conifer plantations were closing canopy at the

Figure 2 Decline in UK emissions of pollutant sulphur since 1970 in response to emission control and predicted future trend.



time when pollutant emissions were peaking in the late 1960s and early 1970s, resulting in maximum pollutant scavenging. UK pollutant emissions have declined substantially since then (1970), by 94% for sulphur dioxide, 58% for oxides of nitrogen and 21% for ammonia by 2010 (RoTAP, 2012). They are also set to continue to fall sharply, with sulphur deposition predicted to decline by 47%, nitrous oxides by 32% and ammonia by 16% by 2020, relative to 2005 (Kernan *et al*., 2010). This means that, by the time a newly restocked forest closes canopy and scavenging becomes significant as a mechanism, pollutant levels are likely to have reduced by around 90% from peak levels for sulphur and by around two-thirds for nitrous oxide.

A recent modelling study by Helliwell *et al*. (2011) predicted that the scavenging mechanism is likely to become increasingly marginal in the future. As the magnitude of the effect declines in absolute terms, the risk of this causing or exacerbating critical loads exceedance becomes very low. Consequently, restocking is increasingly unlikely to delay the recovery of acidified waters.

The main caveat to the expected low risk of restocking contributing to future acidification concerns the role of nitrogen. Forest systems are generally very effective at locking up most of the incoming nitrogen from the atmosphere, either in biomass or forest soils. However, work by Stevens *et al*. (1994) found evidence of ageing (>45 years) conifer forests in areas subject to high nitrogen deposition becoming saturated with nitrogen and starting to leak nitrate. Forests tend to have a high nitrogen demand during the early period of growth, when large amounts are required to form foliage (with a relatively low carbon to nitrogen ratio), which results in low levels of nitrate leaching. After canopy closure, less nitrogen is required to form stem wood (which has a very high carbon to nitrogen ratio) so that demand can be met, as for base cations, by recycling from litterfall. At this stage, forests are more susceptible to nitrogen 'saturation', where a proportion of deposited nitrogen is no longer retained by the ecosystem, and nitrate leaching can occur. The mineralisation of accumulated litter in mature stands may also lead to rates of nitrate leaching that exceed deposition inputs (Stevens *et al*., 1994). These factors are thought to be responsible for conifer forest streams in areas of high nitrogen deposition often having two to three times the concentration of nitrate found in comparable moorland streams.

Restocking provides an opportunity to redesign a forest to reduce its scope for contributing to acidification. In particular, the UK Forestry Standard, and sustainable forest management, are driving the increased diversification of upland conifer forests, in terms of forest age, type, species and structure (Forestry Commission, 2011c). First rotation forests were typically uniform-aged, conifer monocultures, which helped to maximise pollutant scavenging and the risk of nitrate leaching at a forest and catchment scale. Forest management plans introduced in the last decade aim to break up these monocultures by introducing more open space, broadleaved woodland, different conifer species and, perhaps most importantly, a greater range of forest age. Since pollutant scavenging only becomes important after canopy closure at around 15 years of age and conifers are typically managed on a 35–45-year rotation, more than a third of the area of a mixed aged forest would not exert a significant scavenging effect at any given time. This factor, combined with the conversion of at least 10% of the conifer area to open ground, represents a marked reduction in pollutant scavenging capability (Forestry Commission, 2011c). While the increased extent of forest edge resulting from these design changes will partly counter the reduction in pollutant scavenging, the overall effect of this is expected to be relatively small in view of the typical size of forest blocks (see the section above on planting scale and design).

Another important change is the move to smaller-scale clearfells and alternative practices in future rotations, as well as the conversion of at least 5% of conifer area to broadleaves and the establishment of riparian woodland buffer zones. Reducing the area of felling limits the potential release of nitrate at the catchment scale and makes it more likely that any leached will be retained by downslope stands. A shift to continuous cover forestry or low impact silvicultural systems will also help in this regard, as will the shift to more broadleaves. While these management options are not expected to reduce pollutant scavenging, they could help to reduce the risk of nitrate release from the soil and thereby effective nitrogen deposition. The riparian buffer zone provides further scope for nutrient retention as long as it is not bypassed by forest drains, as well as improving riparian and aquatic habitats through leaf litter and dead wood inputs, and the provision of shade. Riparian woodland buffers have been shown to benefit fish populations and thus can be expected to aid the biological recovery of acidified waters as chemical conditions improve (Broadmeadow and Nisbet, 2004).

Identifying vulnerable areas

Acidification occurs where the inputs of acid pollutants exceeds the buffering capacity of the soils and the underlying rocks through which water drains before entering surface waters. The most acidified areas in the UK are in uplands where base-poor, slow weathering soils and rocks have been subject to high pollutant inputs. This includes parts of central and southwest Scotland, Cumbria, the Pennines, Wales and the Mourne Mountains in Northern Ireland.

It is clearly important to be able to identify areas and waters that are susceptible to acidification so that appropriate controls and measures can be put in place to protect these from any forestry effect. The more precise the identification, the better targeted and effective the measures can be. Mapping of susceptible areas has greatly improved over time with developments in methodology and monitoring, although it remains constrained by a lack of water chemistry data within upland areas.

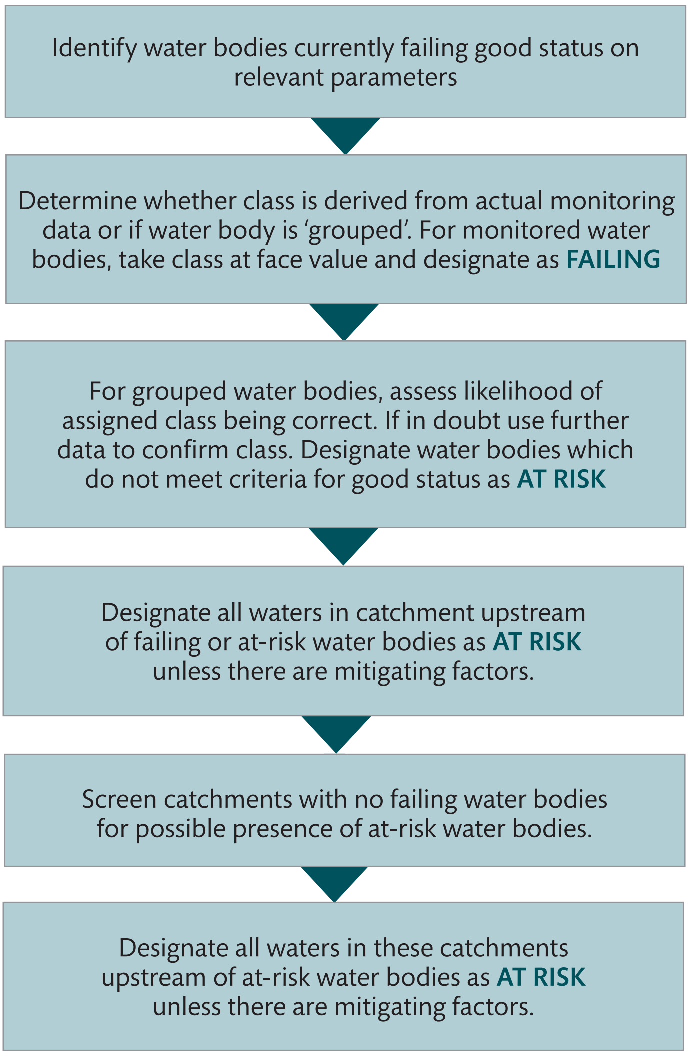
Initially, maps simply characterised acid sensitivity based on the distribution of known acidic soil and rock types, modified by land-use activities such as the liming of agricultural soils (Hornung *et al*., 1995). The introduction of the critical loads concept in the 1980s allowed a more quantitative approach to be adopted by directly comparing the available buffering in sampled waters with acid deposition loads. However, this was still hampered by the lack of water chemistry data such that the resulting critical loads exceedance map used by the third edition of the *Forests & water guidelines* (Forestry Commission, 1993) to define susceptible areas was limited to a coarse 10 km scale. Its indicative nature also meant that all adjacent squares had to be considered potentially at risk.

Improvements in chemical and biological monitoring over the past 10 years, driven by the EU Water Framework Directive, has allowed the acid sensitivity of all UK water bodies to be assessed, and therefore the target area to be better defined. This is now based on those river and lake water bodies in the UK measured by the water regulatory authorities as currently failing, or at risk of failing, Good Ecological Status due to acidification (Forestry Commission, 2011a). Failing water bodies are those where regular chemical (pH for rivers and ANC for lakes) or biological (acid score based on benthic macroinvertebrates) measurements show that conditions fail to meet defined environmental quality standards.

At-risk water bodies are where no direct measurements are available but there are reasons to suggest local waters might be vulnerable (e.g. due to proximity to failing water bodies and nature of soils and geology). Water bodies measured as not failing may also be considered to be at risk where information indicates that local waters upstream of measurement points could be susceptible; this is important, because rivers tend to become less acid sensitive further downstream, and the relatively large catchment scale at which Water Framework Directive assessments are undertaken can fail to detect acidification in smaller headwaters.

The methodology used to define failing or at-risk water bodies is described by Doughty (2011) and illustrated in Figure 3.

Figure 3 Decision tree for identifying waters that are failing or at risk of failing Good Ecological Status due to acidification (from Doughty, 2011).



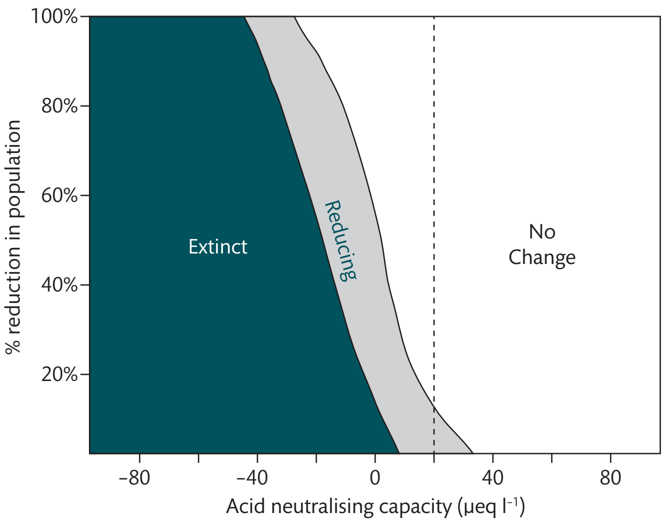
Critical loads approach

The critical loads approach provides a way of quantifying the available buffering within a catchment and determining whether this is sufficient to neutralise acid inputs. A critical load is formally defined as 'the highest deposition of acidifying compounds that will not cause chemical changes leading to long-term harmful effects on the ecosystem structure and function' (Nilsson and Grennfelt, 1988). In the context of forestry, it is used to identify surface waters where the catchment supply of base cations required to protect fish is exceeded by acid deposition, and therefore where any additional pollutant capture by forestry is likely to cause acidification and damage local ecology. It is applied to individual sub-catchments subject to new planting or restocking proposals within at risk and/or failing water bodies.

There are two types of critical loads models, steady state and dynamic. Steady-state models use a simple 'mass balance' calculation to determine whether for a given level of acid deposition catchments are able to supply enough base cations or ANC to maintain ANC in surface waters above a defined threshold for protecting freshwater life. This type of model cannot predict the timescale for reaching steady-state or equilibrium conditions, which could take many decades. Consequently, it is often the case that present-day observed chemical and biological conditions do not match steady-state model predictions since there will be a time lag for soils and waters to adjust and re-equilibrate to the current level of acid deposition. New planting or restocking in the interim would not be expected to halt the recovery process provided that the scavenging effect does not cause future deposition to exceed the critical load. The continuing decline in acid deposition and improvements in forest design and management practices should make critical load exceedance increasingly unlikely, as long as nitrogen saturation does not become an issue.

A core element of the critical load models is the choice of ANC threshold, the value of which depends on the element of ecology selected for protection. The brown trout is usually the species chosen and measured 'dose–response curves' are used to identify an appropriate value (Figure 4; Lien *et al*., 1996). Initially an annual mean of ANC 0 was recommended by the UK Critical Loads Advisory Group for use in critical load applications, equating to a 50% probability of the trout population being protected from damage. This was subsequently increased to ANC 20, giving a 90% probability of protection. More recently, Malcolm *et al*. (2014b) have questioned the reliability of ANC dose–response curves and recommended the use of a higher threshold of ANC 40 for meeting Good Ecological Status. However, achieving this relatively high value would pose problems for some naturally acidic waters that would originally have had a mean ANC below 40 μeq l^{-1}, making Good Ecological Status unattainable.

Figure 4 Percent reduction in brown trout population status in relation to ANC concentration based on a 1000-lake survey in Norway (from Lien *et al*., 1996).

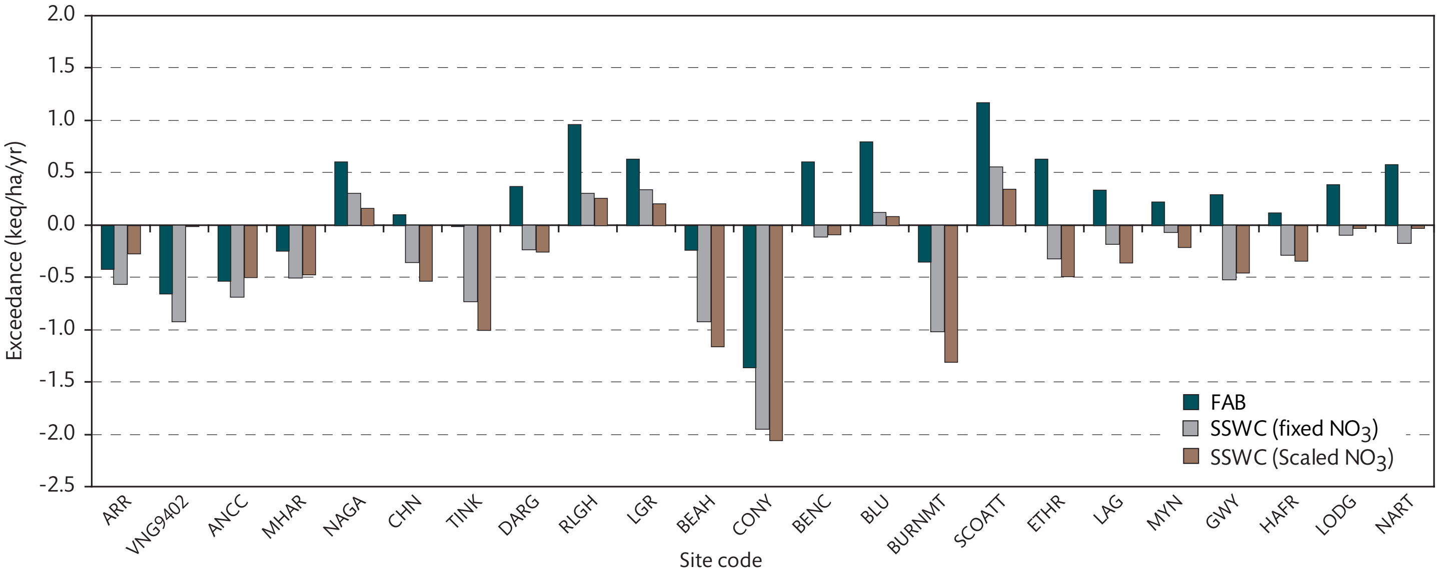


A number of steady-state models are available but the two most commonly used are the steady-state water chemistry (SSWC) model (Henriksen *et al*., 1992) and the first-order acidity balance (FAB) model (Posch *et al*., 1997). The SSWC or 'Henriksen' model is the simpler of the two. It assumes that most nitrogen deposition will continue to be retained within catchments and thus nitrogen processing can be treated as a 'black box'. The small portion of nitrogen that usually appears in surface waters is treated as the effective rate of nitrogen deposition, and added to that of sulphur to determine whether the critical load is exceeded. In contrast, FAB undertakes a more detailed mass balance calculation of all nitrogen sources and sinks within catchments, and any nitrogen input in excess of these long-term sinks is assumed to be leached to surface waters as nitrate and to contribute to the acid load. FAB's ability to handle nitrogen processes and generate separate critical load functions for sulphur and nitrogen has led to it being the preferred choice in Europe for critical load applications to inform emission control policy.

The main dynamic model is MAGIC (Model of Acidification of Groundwater in Catchments), which is a process-orientated model that is able to predict the response time of acid deposition effects on soil and surface water chemistry (Cosby *et al*., 2001). It consists of a series of soil solution equilibrium and mass balance equations, including nitrogen dynamics that embrace the concept of nitrogen saturation. The model can operate at various time-steps and look both backwards and into the future. Work continues to develop the model and to improve its representation of acidification and forest processes, as well as to allow the degree of uncertainty in predictions to be quantified. In particular, recent versions of the model have focused on providing a more realistic representation of nitrogen cycling processes (Oulehle *et al*., 2012), and thus a better prediction of the future extent and trajectory of nitrate leaching.

Kernan *et al*. (2010) compared the predictions of all three of the above models using the 22 surface water sites in the UK Acid Waters Monitoring Network (UKAWMN). They found that there was relatively close agreement between the predicted endpoints of the SSWC and MAGIC models, with all but five sites expected to recover by 2020. The FAB model was more conservative and predicted that 15 sites would remain impacted (Figure 5). This discrepancy resulted from differences in the way that MAGIC and FAB handle nitrogen processes, with FAB predicting a greater extent of nitrogen saturation and nitrate leaching at an undefined endpoint. MAGIC simulations may also predict a similar endpoint if run far enough into the future, but suggest that it may take centuries to reach this point under current

Figure 5 Comparison of critical load exceedance predictions for UKAWMN sites for 2020 using worst (FAB), intermediate (SSWC with no change in nitrate leaching) and best case (SSWC with nitrate leaching scaled to deposition) models (from Kernan *et al.* (2010).



deposition loadings, particularly for organic-rich soils. FAB's prediction of more widespread nitrogen saturation is generally at odds with current observed trends and MAGIC-modelled trajectories on the timescale of the next forest rotation, resulting in the model's reputation for presenting a 'worst-case' scenario (Kernan *et al*., 2010).

Current guidance (Forestry Commission, 2014) favours use of the SSWC model for critical load assessments for new planting and restocking proposals. This is based partly on ease of application but also on the models' relative performance. The additional process representation by FAB and MAGIC make these more data hungry and uncertain as site data are usually lacking for model parameters, requiring the use of default, 'lumped' catchment values derived from the scientific literature.

The SSWC model's assumption that catchment uptake of nitrogen will remain constant in the future is considered to be too conservative for new planting proposals. This is based on catchment studies showing that streams draining conifer forest typically have two or three times higher nitrate concentrations than their moorland counterparts in areas receiving high nitrogen deposition (Stevens *et al*., 1997). Although there is some evidence of this difference declining with emission reductions (Kernan *et al*., 2010), a nitrate multiplier of times three is applied to account for the potential effect within areas at risk of nitrogen saturation, adjusted for the planned scale of forest cover in the catchment. The multiplier is reduced by half for broadleaved planting to reflect the perceived greater ability of broadleaved woodland soils to retain nitrogen (Tipping *et al*., 2012). No multiplier is applied to restocking proposals since any land-use change effect has already taken place, and declining emissions and ongoing improvements to forest design are expected to reduce nitrate concentrations in waters draining existing forests, which it is assumed will offset any existing nitrogen saturation.

The standard application of the SSWC model uses at least a year's worth of monthly sampling in order to calculate the critical load. However, for reasons of timing and cost it can be problematic to wait for 12 months or more to generate an annual mean chemistry to use in the calculation. Instead, the calculation is based on the collection of one or more high flow samples from affected surface waters. This is justified by waters usually being at their most acidic during high flow events when drainage is mainly through upper soil layers. As acidity changes rapidly with increasing flow and the relationship can vary between seasons, it is important to collect samples during 'spate' conditions, preferably between January and March when biological activity is lowest and therefore nitrate leaching is greatest. To compensate in part for sampling under such conditions, a lower ANC value of 0 was previously selected as the critical threshold for protecting fish. Studies show that the difference between high flow and mean ANC is usually much greater than 20 μeq l^{-1}, suggesting that a threshold of ANC 0 for high flow is more precautionary than the ANC 20 applied to mean chemistry (Nisbet *et al*., 2007).

Current guidance (Forestry Commission, 2014) adopts a more precautionary approach by increasing the high flow threshold from ANC 0 to 20. An alternative option may be to calculate the critical load using a small number of medium or low flow samples by relating the chemistry to a local analogue, long-term monitoring site. This would provide an estimate of mean chemistry and overcome the practical difficulties of collecting a high flow sample. In such circumstances, it would be appropriate to apply a threshold of ANC 40 based on the work of Malcolm *et al*. (2014).

Research and monitoring

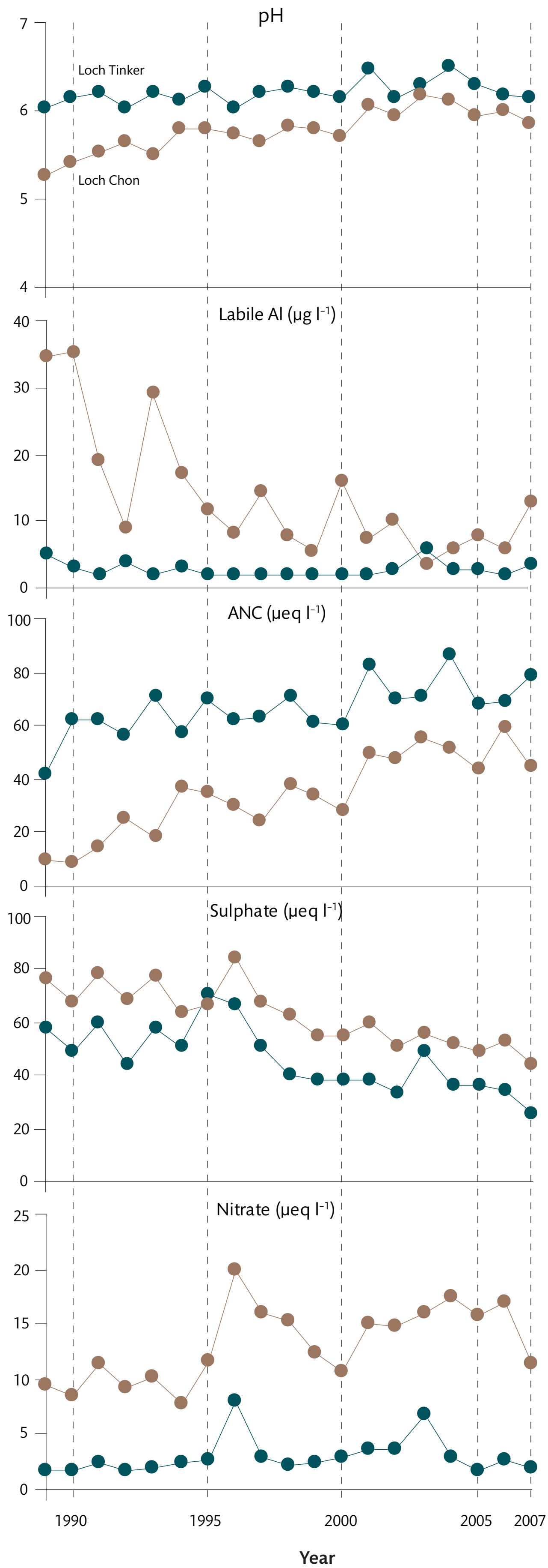
Several long-term studies are in place across the UK to measure the response of acidified waters to emission control and to look at the role of land use in acidification and recovery processes. The data from these are key to aiding our understanding of the impacts of acid deposition on catchment soils and waters, interactions with land use and climate change, and for testing model predictions. The most comprehensive study is the UK Acid Waters Monitoring Network (UKAWMN; now known as the UK Uplands Water Monitoring Network), comprising 22 river and lake sites distributed across the uplands of Scotland, England, Wales and Northern Ireland. This was established in 1988 and includes 5 forest and 17 moorland sites. The 20-year report published by Kernan *et al*. (2010) found clear evidence of both chemical and biological recovery in acidified lakes and streams across all affected regions. In response to lower emissions, recovery is predicted to continue to 2020 and beyond but is thought unlikely to return waters to their original pre-acidification status of before the industrial revolution. There is also continuing uncertainty about how the recovery process will be affected by future nitrogen deposition and climate change.

In terms of the role of forestry, Kernan *et al*. (2010) found that the forest sites in the network remain more impacted than their moorland counterparts, with higher pollutant sulphate, nitrate and stream acidity (aluminium and/or pH). However, they are recovering in line with expectations and show evidence of sulphate, aluminium and ANC concentrations converging strongly as a result of more rapid declines in the acidified forested sites. At one of the five forested sites, Loch Chon in central Scotland, aluminium concentrations have declined to background levels in recent years, and the combined effect of this and the consequent strong pH response has led to substantial ecological improvement (Figure 6). Biological recovery at the other four sites remains weak due to insufficient declines in aluminium and/or increases in pH to date.

There has been a variable response in nitrate concentrations across the forested sites. This is thought to be due to the influence of forest management, with shorter-term nitrate losses from those subject to clearfelling contributing to stable or rising trends, while the strong nitrate uptake by replanted forests enhanced declining trends in others. Base cation concentrations have tended to remain higher within the forested sites, leading Kernan *et al*. (2010) to suggest that base cation depletion by tree growth and timber harvesting has not been a major factor influencing site recovery to date.

An application of the MAGIC model to the 20-year dataset predicted that four of the five forested sites would reach the target of ANC 20 by 2020. The model also predicted that nitrate

Figure 6 Comparison of mean annual chemistry time-series between the moorland site of Loch Tinker and the neighbouring forested site of Loch Chon in the Scottish Trossachs (from Kernan *et al.* (2010).



concentrations would only rise by a small margin in the future (at least to 2100), base cation removal would have a negligible impact on acidification, and, perhaps most importantly, deforestation as a potential mitigation option is unlikely to significantly alter the path to recovery (Kernan *et al*, 2010). The predictions of recovery matched those of the SSWC model but not FAB, which suggested that nitrogen saturation would confound recovery and lead to four of the five sites failing to achieve ANC 20. However, recent developments of MAGIC applied to forest sites in the Czech Republic, including a better representation of microbial nitrogen cycling, suggest that the nitrogen leaching response may be more dynamic over time (Oulehle *et al*., 2012). Further model development and testing is required to assess the longer-term susceptibility of UK forests to increased nitrate leaching. It remains unlikely though that the FAB 'steady-state' levels of nitrate leaching will be attained within the timescale of at least the next forest rotation.

A more recent application of the MAGIC model to southwest Scotland supports the findings of Kernan *et al*. (2010). Helliwell *et al*. (2011) found that the forest scavenging effect would only make a small contribution to future pollutant loads due to the reduction in atmospheric concentrations. The difference in predicted ANC concentrations between 'forest' and 'no-forest' scenarios was considered not to be significant in terms of achieving ANC thresholds. This led the authors to conclude that future changes in forest cover were likely to have a relatively minor effect on the recovery process.

The results from other long-term monitoring studies are largely in line with those of the UKAWMN. This includes the joint Forestry Commission and Environment Agency network of 10 forest and 2 moorland catchment streams across upland Wales, which was established in 1991 to supplement the UKAWMN. All sites display elements of chemical recovery, including consistent declines in non-marine sulphate concentrations and rises in pH, but no significant change in aluminium.

Also in Wales, the Centre for Ecology and Hydrology has been operating two sets of paired forest and moorland catchment streams at Plynlimon and Beddgelert since the 1980s (Neal *et al*., 2001; Reynolds *et al*., 2004). Findings are generally in line with those described elsewhere, with similar chemical recovery trends in forest and moorland streams, but continuing offsets for some parameters, including aluminium, sulphate and nitrate concentrations. At Beddgelert, it appears that nitrate concentrations in the forest stream have declined towards those of the moorland, from approximately six to around two times higher. This change has occurred through a period of declining nitrogen deposition, but also growth of the second rotation forest, so the relative importance of these two drivers is hard to distinguish. At Plynlimon, the offset for nitrogen has remained more stable, with nitrate leaching from forest streams approximately double that from the moorland sites. An experimental felling study within part of one catchment showed a large nitrate peak extending over several years, but little evidence of an associated increase in acidity (Neal *et al*., 2004).

Another key study involves eight catchment streams varying in percentage forest cover and forest age at Loch Ard in central Scotland. Sampling began in 1976 and a recent analysis of the data by Malcolm *et al*. (2014a) found strong evidence of both chemical and biological recovery in line with reductions in acid deposition. Reductions in non-marine sulphate and aluminium concentrations and increases in pH and ANC were greatest for the sites with the strongest forestry influence. However, despite these improvements, streams dominated by forestry remain more acid than those with a modest forestry influence (<30% cover and 25-30 years age) or with a moorland cover. Similar to the UKAWMN, this shows that catchments planted extensively with conifer forest in the 1950s and 60s, and closing canopy during the period when atmospheric deposition was at its highest, continue to display a legacy effect. The timescale for achieving a chemical status capable of supporting acid-sensitive species may therefore be longer, depending on the rate of convergence. This matches the findings of repeated regional surveys of streams and lochs in Galloway in southwest Scotland, which show evidence of chemical recovery but a relic forest effect, particularly in sites dominated by forestry (Dunford *et al*., 2012).

Ormerod and Durance (2009) also found evidence of chemical recovery in acidified streams at Llyn Brianne in south Wales. Monitoring of 14 forest and moorland catchments began in 1981. An analysis of the first 25 years of data to 2005 showed similar levels of decline in hydrogen ion concentration between both sets of streams. However, as with the other studies, the lower initial pH of the forested streams mean that they remain more acid with higher aluminum concentrations, which is delaying the recovery of acid-sensitive macroinvertebrate species. Similarly to Malcolm *et al*. (2014a), the decline in acidity appears to be greater in older than in younger forested catchments, reflecting the more acidified condition of the former. Wetter winters were found to influence time trends, indicating that climate change could affect recovery processes. While minimum pH levels are also rising, acid episodes continue to act as a break on biological recovery in both acidified forest and moorland streams.

Maintaining these long-term studies will be key to demonstrating whether the recovery process continues in line with MAGIC and SSWC model predictions, closing the gap between forest and moorland sites and achieving target ANC, or alternatively whether recovery is halted or even reversed by increasing nitrogen saturation (as predicted by the FAB model) or offset by climate change.

Promoting recovery

Securing the planned reductions in acid pollutant emissions remains the priority for tackling the primary driver of acidification. Achieving these reductions should help many acidified water bodies to meet the target ANC, although not necessarily Good Water Status, depending on the final selection of environmental quality standards. Modelling suggests that the dramatic improvements in air quality in recent decades will also reduce forestry's contribution to acidification to a small margin, although it may take longer for forested sites to recover from their more acidified condition compared to moorland sites, due to the legacy of higher pollutant inputs because of scavenging in the past (Kernan *et al*., 2010; Helliwell *et al*., 2011). Action to remove existing forest cover or prevent new planting is unlikely to have a marked effect in promoting chemical recovery in many cases.

For the most acid sensitive of surface waters, there is a risk that these will remain impacted even by the reduced levels of acid deposition, requiring continued restrictions on new planting and forest restocking. There is also continuing uncertainty about the risk of nitrogen saturation and the impact of climate change. These issues will need to be kept under review. A switch from coniferous to broadleaved species might help forest soils to retain more nitrogen and reduce nitrate leaching (Tipping *et al*., 2012), as would converting conifer stands to continuous cover forestry or low impact silvicultural systems.

There are a range of measures that can be used to promote the recovery of acidified waters within forested catchments, involving changes to forest design and improvements in management practices. One of the most important is clearing densely-shading conifers along streamsides. While there is limited evidence that this affects stream chemistry, studies show that the physical improvements to aquatic and riparian habitats can significantly increase invertebrate abundance and numbers of trout where water quality is suitable (Broadmeadow and Nisbet, 2004). Consequently, targeted clearance of riparian conifer stands casting heavy shade could aid upstream fish migration and the biological recovery of streams showing chemical improvement in response to ongoing emission reductions. However, the full benefit will take a number of years to develop and depend on active management of the riparian zone to control conifer regeneration and establish an open canopy of native broadleaved woodland.

Liming is sometimes advocated to promote recovery. However, a recent systematic literature review found that, while on average liming increased the abundance and richness of acid-sensitive invertebrates and increased overall fish abundance, the benefits were variable and not guaranteed (Mant *et al*., 2011). The authors calculated that there was an 18% probability of liming reducing fish abundance, no overall effect on trout abundance where salmon were also present (the mean effect was negative but not significant), and an indication of an overall negative effect on invertebrate abundance. Weaknesses in the experimental design of many of the reviewed studies, including a lack of control sites, limited confidence in the results. The significant risk of a range of ecologically negative impacts makes it difficult to justify liming when natural recovery is under way, albeit slowly.

Conclusions

The primary mechanism responsible for a forestry acidification effect is the ability of forest canopies to capture more sulphur and nitrogen pollutants from the atmosphere than shorter types of vegetation. Base cation uptake and removal generally exerts a small acidification effect, except where soils are extremely base poor or where whole-tree harvesting is practised. Pollutant scavenging is thought to have peaked in the 1970s when emissions were greatest and the planting of extensive conifer plantations within acid-sensitive upland areas in the 1950s and early 1960s reached canopy closure. This led to surface waters draining catchments dominated by forestry being more acidic, with higher concentrations of non-marine sulphate, nitrate, aluminium and/or hydrogen (lower pH).

The introduction of emission control policies in the 1980s has achieved major improvements in air quality. This has led to marked chemical recovery and increasing evidence of biological recovery in acidified lakes and streams across all affected regions of the UK. Recovery is predicted to continue to 2020 and beyond but is thought unlikely to return waters to their original pre-acidification status. There is also uncertainty about how the recovery process will be affected by future nitrogen deposition and climate change.

Monitoring studies show forest sites to be recovering in line with their moorland counterparts, with some evidence of convergence in chemistry. Despite this, forest streams remain more impacted, indicating that the timescale for recovery may take longer. Modelling suggests that the improvements in air quality will reduce forestry's contribution to acidification to a small margin, such that action to remove existing forest cover or prevent new planting is unlikely to be required to achieve chemical recovery in many cases. However, there is a risk that the most acid sensitive of surface waters will remain impacted by the reduced levels of acid deposition, requiring continued restrictions on new planting and forest restocking. Appropriate controls and measures are in place, including catchment-based critical load assessments and site impact assessments to protect these from any potential forestry effect. Continued monitoring is essential to demonstrate whether existing measures remain fit for purpose and guide the need for future revisions to guidance on good practice.

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