UCL ENVIRONMENTAL CHANGE RESEARCH CENTRE

A Review of Liming as a Technique for Protecting Salmonid Fish Populations in Acidified Surface Waters

Report for the West Country Rivers Trust

ECRC Research Report Number 156

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1. Executive Summary

Liming is a common technique that has been used in many countries to raise the alkalinity of acidified surface waters and alleviate some of the damaging effects of acidification on salmonid fish populations.

The most common liming substance used is calcite, a calcium carbonate compound that is relatively inexpensive, available in different particle sizes and dissolves relatively quickly.

It can be applied directly to streams or lakes or it can be applied to catchment soils. When applied to catchment soils its effect can be long-lasting but it can cause significant damage to those catchment plant and animal communities that are naturally adapted to acidic conditions. When applied directly to surface waters its effect can be immediate but applications need to be continuous or frequently repeated to counter downstream dilution and loss.

For streams the most effective method is to use an automatic doser controlled by pH-measuring sensors upstream and downstream of the doser to enable the exact quantity of lime needed to be added to the water body. Although effective this is an expensive method and one that needs to be maintained continuously for several years until the critical load exceedance has been eliminated. An alternative or complementary method is partial catchment liming by targeting water sources and selected wetlands to minimise damage to catchment vegetation.

Liming can be very effective in restoring and protecting salmonid fish populations, but if over-applied it can lead to unwanted increases in alkalinity and productivity that may produce symptoms of eutrophication and unwanted changes in the composition of plant and algal communities downstream.

Before a decision to lime is made it is important consider all possible uncertainties and specifically to have:

- (i) a clear justification, including a good reason for intervening before natural recovery has had time to occur;
- (ii) a good biological and ecological understanding of the target species and the ecosystem being modified;
- (iii) a knowledge of the target area, especially its hydrology;
- (iv) an understanding of the collateral damage to other plant and animal communities that might occur;
- (v) an appreciation of the length of time needed for the treatment to be applied;
- (vi) an awareness of the long-term feasibility of the program;
- (vii) a commitment to assessing the impact of the treatment through long-term monitoring.

2. Introduction

Anthropogenic acidification of surface waters from emissions of SO_2 and NO_x has since the 1970s been widely recognised as one of the major environmental problems in Europe and North America (Mant *et al.* 2011; Clair & Hindar 2005; Menz & Seip 2004). Acid rain impacts aquatic ecosystems by lowering pH and can increase labile aluminium concentrations beyond lethal toxic thresholds for organisms, leading to biodiversity loss and profound alteration of community structure and ecosystem processes (Mant *et al.* 2011; Schindler *et al.* 1985). The degree of damage to ecosystems by acidification is determined by the sensitivity of bedrock and soils in a region and the amount of acid deposition received through time (Clair & Hindar 2005). Southern Scandinavia is one of the European areas most heavily affected by acidification. The region has slowly weathering bedrock and has been exposed to long-distance transported sulphur and nitrogen emissions emanating from fossil fuel combustion in heavily industrialised countries, principally the UK, upwind. Because of these factors brown trout (*Salmo trutta*) populations have been lost from large numbers of lakes and streams in Southern Norway and South-west Sweden, and, in Norway, Atlantic salmon (*Salmo salar*) populations have been lost from about 20 rivers (Hesthagen & Larsen 2003). Upland lakes and streams in the UK have been even more severely affected following more than a century of acid deposition most clearly seen from the history of air pollution and its ecological effects recorded in lake sediments (Battarbee *et al.* 1988).

Since the early 1980s rigorous acid emission controls have been successfully introduced across Europe and North America resulting in significant reductions in acid deposition and improvements in surface water chemistry (Mant *et al.* 2011; Reynolds *et al.* 2004, Evans *et al.* 2001, Stoddard *et al.* 1999, Monteith *et al.* in press). Nonetheless, sulphate levels in surface waters, although declining, remain higher than background, while NO_x emission levels remain in most cases unchanged or if reduced, levels are still on a higher base than those achieved for SO_2 emissions (Mant *et al.* 2011; Fowler *et al.* 2007). At present, levels of nitrate within surface waters remain relatively high in many regions (Mant *et al.* 2011; Reynolds *et al.* 2004, Monteith *et al.*, in press). Recovery is a slow process and problems of acidification persist.

In an attempt to protect fish populations and accelerate ecosystem recovery from acidification many countries have, to a greater or lesser extent, used some form of liming or other buffering substance to increase pH (Clair & Hindar 2005; Sandøy & Langåker 2001; Henriksson & Brodin 1995). The liming of lakes, streams, and rivers for this purpose has been widely studied experimentally in North America and Europe for over 20 years. There is an imposing amount of literature describing the results of experimental liming, as well as a number of reviews of large-scale European projects, including the UK (e.g. Ormerod & Durance 2009, Henrikson & Brodin 1995, Howells & Dalziel 1992, Edwards *et al.* 1990). However, now that acidified lakes and streams in the UK are beginning to recover as acid deposition decreases there is less justification for interventions such as liming that, whilst protecting fish populations, may have negative consequences for other aspects of upland freshwater ecosystems.

Nevertheless, in this report, we describe the standard liming compounds and application techniques that have been used; we review their potential effects on aquatic ecosystems and highlight some of the most relevant liming studies that have taken place in the UK. For specific details of liming methods we refer the reader to the review of Olem (1991).

3. Common Buffering Compounds:

The following buffering compounds have been used in acidification mitigation (Table 1) (Clair & Hindar 2005, Olem 1991):

1. **Calcite** ($CaCO_3$**):** the most commonly used. Due to the extent of its general use in agriculture, calcite is easily available, relatively inexpensive, and has been well studied in the past. It can be purchased in different particle sizes, which makes it very useful for the different application methods (see next section).

2. **Dolomite** ($CaMg(CO_3)_2$): closely resembles calcite in its characteristics, has slightly higher buffering capacity but dissolves more slowly in acid water. It is also commonly available and has been used in experimental studies. It is mostly applied on catchment soils rather than as an addition to water itself.

3. **Sodium carbonate** (soda ash, Na_2CO_3) and **sodium bicarbonate** (baking soda, $NaHCO_3$): both materials are effective buffering agents and commonly applied in water treatment facilities. They are, however, considerably more expensive than calcium-based buffers and have not been tested extensively in field situations.

4. **Mineral oxides** (calcium and calcium-magnesium oxides, CaO and CaO-MgO, quicklime or cementkiln dust (CKD)): are the byproducts of cement manufacture. They are mostly used as buffering agents in water treatment plants. Mineral oxides are however, difficult to store and apply safely in the field (Olem 1991). Studies in Sweden (Dickson & Brodin 1995) and Norway (Baalsrud *et al.* 1985) specifically council against their use in the field, due to high potential of injury to staff and damage to machinery. Careless use or accidental releases can also "induce harmfully high pH in the water" at the local scale (Dickson & Brodin 1995).

5. **Oxides**: Commonly (especial NaOH) used in water and wastewater treatment. Due to their caustic nature is not recommended for applications in natural situations (Olem 1991). They are especially difficult to use under rainy conditions when hydration can occur in equipment or in temporary open storage. However, Calcium (slaked lime, $Ca(OH_2)$), calcium-magnesium ($(CaMg(OH)_4)$), and sodium hydroxide (caustic soda, NaOH) are less likely to react with moisture than the mineral oxides. Yet, they can still potentially cause high pH shock in waters when over applied.

6. **Silicates**: slag lime, a waste product of metal smelting, has been used experimentally in the 1970s and 1980s but has been rejected due to its high metal content and detrimental effects on soils and aquatic populations (Alenäs *et al.* 1991). Olivene (Mg_2SiO_4) has also been tested in a Swedish catchment application, but because of its slow dissolution its application is not practicable (Wilander *et al.* 1995). Crushed wollastonite ($CaSiO_3$) has recently been applied in one of the Hubbard Brook catchments, New Hampshire, USA, by Likens *et al.* (2004) with improvements in water pH, ANC, and base cations only during base flow conditions. Silicates tend to dissolve much more slowly than carbonate minerals and thus less used for rapid action. Sodium silicate ($Na_2O{\cdot}SiO_2$) forms silica lye ($Si(OH)_4$) in water, which binds strongly to aluminium. Its application is still mostly experimental but results are promising where the aim is to detoxify Al faster than by the use of calcite Birchall *et al.* (1989).

Table 1 Main compounds assessed for acidification mitigation studies. Theoretical neutralization equivalents of are given in relation to $CaCO_3$ (from Olem 1991).

Common name	Formula	% Neutralisation equivalent
Limestone	$CaCO_3$	100
Dolomite	$CaMg(CO_3)_2$	109
Sodium carbonate	Na_2CO_3	94
Sodium bicarbonate	$NaHCO_3$	119
Calcined lime (CKD)	CaO	179
Calcined dolomite	CaO-MgO	207
Hydrated lime	$Ca(OH)_2$	135
Caustic soda	NaOH	125

4. Liming Methods

The details of the different liming methods have been set out in detail in Olem (1991), Donnelly *et al.* (2003), Clair & Hindar (2005) and White (2000). However, a brief overview of the most common techniques described in these review papers (especially Clair & Hindar, 2005) is set out below.

4.1. Direct application of lime to running waters

The application of lime or other buffering materials directly to streams or rivers (commonly referred to as in-stream liming) quickly neutralizes water acidity, resulting in immediate improvements of water pH and ANC. This method titrates running waters systems (streams or rivers) through a doser that distributes slurry made from a known amount of fine-grained limestone powder that is mixed with water. The exact quantity of lime that is necessary to be added into a stream or river is usually controlled by pH measuring sensors upstream and downstream of the doser. This is a highly intuitive approach and widely used in Scandinavia.

Advantages: In-stream lime dosing provides a number of advantages: (i) running water bodies that are known to be imperative for fish migration can be directly targeted and carefully controlled (e.g Bjerknes & Tjomsland 2001); (ii) according to the target fish species, pH and calcium concentrations can be controlled to achieve any specific optimum requirements on a continuous basis; and (iii) in-stream liming is a cost-effective method as it reduces waste of liming agents and maintains continuous dosing to meet the target pH levels.

Shortcomings: There are some limitations to in-stream liming that need to be accounted for: (i) the necessary equipment for dosing operations is expensive. There is a requirement for continued maintenance and a constant provision of lime. According to the objectives of the project and its geographical location, dosers may need to be operated year round increasing the amount of equipment maintenance and risk of damage. Thus, if funds are restricted, only a limited number of dosers may be able to be accurately used; (ii) although in-stream liming provides an immediate buffering effect, its long-term effects are strongly dependent on the recovery of the associated catchment soils and the supply base cations and alkalinity suitable for sustainable “natural” populations. Therefore in-stream liming projects may have to be assessed for many decades; and (iii) only waters downstream of the liming site will be affected by water chemistry changes. Consequently, only fish passage and fish habitat at specified distances below the treatment site will benefit. Dilution of limed water by downstream, untreated watercourses will depend on the area drained and the applied dose. It is therefore very important to carefully assess the timing of the project, how many dosers are needed and the extent of funding prior to any installation attempt.

4.2. Application of lime to stream sediments

Another mechanism of reducing acid water conditions in smaller upstream sites is by adding liming substances to their sediments. In contrasting to in-stream liming (addition of a fine powder or slurry), this technique uses gravel-sized particles of lime or dolomite that are placed in the stream sediments and which act as barriers or water filters (Olem 1991). As water passes through these barriers, lime particles are dissolved, releasing bicarbonate and base cations into the system.

Advantages: In addition to improved water chemistry conditions, the application of limestone cobbles in sediments can also act as an important feature for fish spawning. Salmonids generally need coarse sediments for laying eggs in stream and the presence of limestone cobles in the streambed provides temporary improved habitat for the hatching of eggs. Limestone cobbles can also help to ameliorate the interstitial water acidity, which can provide a substantial benefit for egg survival. Buffering material from marine deposits of shell sand that are found in shallow areas along some coasts can also be applied as additional material for this purpose. This is a common practice in Norway, which has resulted in positive effects on brown trout reproduction.

Shortcomings: potential problems associated with the application of limestone cobbles are: (i) the performance of the one-off application barriers tend to decrease rapidly within a few weeks of installation; (ii) sites with large hydrological dynamics are highly prone to lose their barriers, as bed load materials are moved downstream. High dilution rates during high floods may also result in a loss or diminishing of the effectiveness of the barriers; (iii) barriers rapidly become covered with organometallic coatings that can greatly reduce the dissolution of lime particles (Watt *et al.* 1984).

4.3. Lake liming

Another liming method,d related to in-stream dosing is the addition of buffering substances onto lake surfaces. Lake liming involves the direct application of slurry or fine-grained lime onto the surface of a target lake. It can be applied from boats, on ice before melt, or aerially from airplanes or helicopters. This approach is widely used in Scandinavian acid rain mitigation programs (Dickson & Brodin 1995, Sandoy & Romundstad 1995).

Advantages: The main advantages of lake liming are its relatively straightforward application methods and its low costs. The addition of lime to a lake usually brings immediate increases in pH and ANC and a reduction in Al.

Shortcomings: Some of the associated drawbacks of direct lake liming are: (i) liming products can be easily lost, especially in sites where flushing rates are considerably high (Gubala & Driscoll 1991); (ii) liming material can be also lost into lake bottoms unless the target lake has a mean depth greater that 50 m or extremely fine-ground material (mean particle size <0.02 mm) is used (Sverdrup 1985); (iii) acid snowmelt water from the surrounding catchment generally does not mix with the deeper, lime lake waters, forming an acidic layer at the lake surface on its way out of the catchment (Rosseland & Hindar 1988).

4.4. Application of lime to catchments

An alternative approach to direct liming is the addition of buffering substances to the associated catchment of a target water body. This method is usually conducted from helicopters equipped with specialised application equipment, or fixed-wing aircraft with agricultural spreaders or fire fighting capabilities (Olem 1991). The most common approaches for catchment liming are whole-catchment liming, liming only the discharge areas along the streams and wetland liming.

Unlike whole-catchment and discharge area liming, wetland liming applications are restricted by catchment characteristics, as wetlands are mostly found in lowland areas or near lakes and rivers or in depressions underlain by impermeable substrates in fields and forests (Xenopoulos *et al.* 2003). The soils of wetland ecosystems are predominately composed of slowly decomposing plant material, rich in natural organic acids (NOA) that leak into associated waters and provide a rich source of TOC in lakes and streams (Ritchie & Perdue 2003). The NOAs are amphoteric compounds and thus in freshwaters can behave either as acids at pH levels higher than approximately 5.5 or as buffers at lower pH levels (Clair *et al.* 1992). Due to the low pH of wetland draining waters and the good contact between runoff waters, the addition of buffering substances to wetland catchments is commonly used. Liming wetlands enhances the chemical composition of interstitial water and increases Ca^{2+} and pH to levels not normally found in natural wetland conditions (Kvaerner & Kraft 1995).

Advantages: catchment liming can be successful for several reasons: (i) it avoids the problems of poorly mixed water columns of some standing water bodies; (ii) this approach is better for mitigating smaller, remote (poor access) streams and tributaries; (iii) it has low maintenance costs, as liming material needs to be spread at much lower frequency; (iv) there is no need for complex dosing systems that must be intensively maintained; (v) it may help to increase soil base saturation and help soils to retain Al; and lastly (vi) this approach provides more stable variation in water chemistry as is not affected by extreme weather conditions and high flows (Hindar 2005).

Shortcomings: The biggest problems associated with catchment liming are: (i) the application cost as the amount of lime needed will vary depending on soil types and conditions, as well as water flow paths and spreading area; and (ii) the damage it can do to naturally acidic terrestrial habitats and associated flora and fauna.

5. Potential Effects of Liming on Aquatic Ecosystems

The effects of liming on biological communities are complex and vary between ecosystem types (lakes, streams, rivers, wetlands) and regions. Further, due to their greater hydrological stability and their accessibility, the biological responses of lakes to liming have been more widely studied than those of rivers or streams (Mant & Pullin 2012; Weatherley 1988). A recent review by Mant *et al.* (2011) now summarises the best available evidence to address this deficiency for fish and invertebrates in running waters.

Liming usually increases pH and alkalinity in acidified waters resulting in short-term improved water quality for aquatic biota (Mant *et al.* 2011). Nonetheless, substantial research has shown that liming-induced community changes are not stable. For instance, strong temporal variability, mediated by a return to an acidified state of the communities when liming is discontinued is commonly observed (Clair & Hindar 2005). Consequently, liming may only provide temporary remediation of acidification, rather than a long-term solution (Mant *et al.* 2011, Clair & Hindar 2005, Weatherley 1988). Moreover, the continued application of lime can be in itself an ecosystem-level disturbance (e.g. Angeler & Goedkoop 2010; Weatherley 1988). This is of particular importance in regions where acidity is attributable to natural causes or where acidified systems are showing signs of natural recovery (e.g. Bishop *et al.* 2001, Schindler 1997). Further, the inputs of acid water from runoff or tributaries into limed systems can also generate a chemical aluminium speciation, the effects of which can potentially be more toxic than those of acid waters (Angeler & Goedkoop 2010; Rosseland *et al.* 1992; Teien *et al.* 2004). Other potential effects of liming include an increase in water turbidity and the inorganic content of particulate matter and the precipitation of dissolved organic carbon, nutrients and trace metals (Angeler & Goedkoop 2010; Wällstedt *et al.* 2008; Kullberg 1987). Acid episodes during rainstorms or snowmelt can also affect sensitive organisms even under limed conditions (Kowalik *et al.* 2007).

Based on the succinct reviews by Mant & Pullin (2012), Mant *et al.* (2011) and Weatherley (1988) we set out below a brief overview of the different impacts of liming in water chemistry and biological groups.

5.1. Liming and water chemistry

A number of studies have been carried out to evaluate the effect of catchment liming on surface water chemistry (Claire & Hindar 2005). Olem (1991) reviewed a number of the earlier studies and most of these produced improvements in pH, ANC, and base cations in waters draining the limed areas. One of the longest data records of water chemistry post whole-catchment liming is from Tjønnstrond in Telemark, southern Norway. This large dataset provides evidence from a 25 ha area that was limed with 0–0.2 mm calcite powder in 1983, approximately 3 t ha^{-1} (Traaen *et al.* 1997). Despite initial amelioration, by 1990 pH had decreased to 5.0 and Al increased to 70 µg L^{-1}. Water chemistry however has been acceptable for brown trout for over 20 years since the initial treatment. In addition to pH, Ca concentrations have also been measured, showing an increase during pH depressions. The reduced acid deposition and increased base saturation of the soils in response to the liming have worked together to maintain and even improve the water quality again since 1990 (Traaen *et al.* 1997).

5.2. Liming and organic matter decomposition

The decomposition of plant detritus and other organic matter is a vital process to the flow of energy and nutrients in aquatic ecosystems (Weatherley 1988). In acidic lakes the rate of decomposition of organic matter is usually low and predominately driven by fungi (Weatherley 1988). The addition of lime and its associated increase of pH generally stimulate bacterial growth resulting in a switch from fungal to bacterial dominance and an associated increase in the rates of decomposition (Weatherley 1988; Gahnstrom *et al.* 1980). Liming also increases the number and activity of aerobic heterotrophic bacteria in the water column (Scheider *et al.*, 1975). Yet, although bacterial densities have been commonly related to increases in pH, positive responses do not always occur when comparing acidic and non-acidic lakes, which in some instances can be related to DOC (Gahnstrom *et al.* 1980). Further, the precipitation of metals after liming to sediments can compete with sediment bacteria for phosphate and hence influence decomposition rates

(Weatherley 1988). However, the potential toxic effects of metals after liming to sediment bacteria are poorly studied.

5.3. Liming and aquatic macrophytes

Macrophytes are key structural drivers of aquatic ecosystems by providing food, spawning ground and refugee from predation for epiphytes, invertebrates and fish (Jeppesen *et al.* 1998). In acidic waters, dominant plant groups include *Juncus*, *Isoetes*, *Lobelia*, bryophytes and blue-green algae (cyanobacteria) (Stokes 1986). In general, macrophyte growth after liming will be inhibited by shading due to high epiphytic growth and phytoplankton blooms (Sand-Jensen & Sondergaard, 1981). Increased transparency in limed humic lakes may stimulate elodeid macrophyte growth such as broadleaf *Potamogeton* species, whereas reduced transparency due to lime-enhanced algal biomass would have the opposite effect (Sand-Jensen & Sondergaard, 1981). Exposed areas normally colonised by *Lobelia* and *Isoetes* in acidic lakes have been reported to decline after liming by competition with other taller growth macrophytes and/or a direct physiological response to the alterations in water chemistry (Roelofs 2002); however, such responses to liming are not generally consistent (Weatherley 1988). Similarly, an occurrence of *Potamogeton natans* after liming has been reported but the processes behind it were not elucidated (Weatherley 1988). The decline of *Juncus bulbosus* on neutralisation can be ascribed to the direct effects of increasing pH, since culture experiments demonstrate its retarded growth at higher pH (Roelofs *et al.* 1984). The great reductions in growth and distribution of *Sphagnum* spp., benthic mats of blue-green algae, and filamentous green algae, notably *Mougeotia* spp., which occur within 1 to 2 years following treatment, have also been attributed to the reduction in H^+ concentrations (Hultberg & Andersson 1982).

5.4. Liming and algae

Phytoplankton is a vital food resource for zooplankton, macroinvertebrates and fish, and largely contributes to the detrital food chain (Weatherley 1988). In streams and rivers, phytoplankton is usually of lesser importance and the epilithon, composed of diatoms, blue-green algae and bacteria, plays a crucial role in the transfer of energy into higher trophic levels (Weatherley 1988). Although, phytoplankton biomass can initially be reduced by liming, research evidence suggests that on average liming increases their diversity within the following few months (Mant & Pullis 2012; Bengtsson *et al.* 1980). The Dinophyceae dominant in acidic lakes are replaced by Chrysophyceae, Chlorophyceae, Cryptophyceae, and Diatomaceae (Weatherley 1988). Phytoplankton diversity increases after liming achieving a circumneutral oligotrophic lake state within 1 to 2 years (Bengtsson *et al.* 1980), although some species may not reappear until later (Hultberg & Andersson, 1982).

The effects of liming on phytoplankton diversity have been commonly attributed to reduce toxicity of Al and H^+ ions, and to an increase in nutrient concentrations (Weatherley 1988). There is evidence for the direct toxic action of Al and an increase of P uptake with increasing pH (Weatherley 1988). Due to the limited availability of P in oligotrophic lakes, competition of phytoplankton with Al complexes for P is an important factor (Hultberg & Andersson, 1982). Increasing grazing pressure by herbivorous zooplankton can also have an indirect effect on phytoplankton biomass in treated lakes (Weatherley 1988). Information regarding liming effects on the epilithon of running waters is scarce. Apart from direct effects of pH and Al, it seems probable that effects of neutralisation on P concentrations will influence primary production in these systems (Weatherley 1988).

5.5. Liming and zooplankton

Zooplankton are a major influence on the community structure of aquatic systems, acting as the intermediate trophic level between phytoplankton and fish (Weatherley 1988). Although some studies have found an initial decrease in zooplankton biomass following neutralisation, in the absence of fish predation liming generally leads to increased abundances of planktonic Rotifera, Cladocera, and Copepoda within a year (Mant & Pullis 2012; Weatherley 1988; Bengtsson *et al.*, 1980). *Bosmina* spp. (Cladocera), which often dominates at low pH, are replaced by other less acid tolerant taxa, such as *Diaphanosoma brachyurum* (Cladocera) (Weatherley 1988). Further, high concentrations of H^+ and Al^{3+} ions are known to be toxic to certain species of zooplankton affecting their ionic regulation (Weatherley 1988). Increases in phytoplankton production must favour herbivorous species of zooplankton, provided species composition of the phytoplankton is suitable.

Much attention has been focused on the role of fish and invertebrate predators (Chaoboridae, Corixidae, Dytiscidae, and others) in controlling the response of zooplankton to liming (Weatherley 1988). In the absence of fish, lime treatments lead to a higher abundance of invertebrate predators, resulting from increased secondary productivity and/or improvement in water quality (Weatherley 1988). Research has also reported an inverse relationship between the abundance of chaoborids and liming, which was enhanced in limed lakes, and the number of cladoceran species and their density (Weatherley 1988). Fish predation may reduce invertebrate predator numbers, allowing establishment or increased density of other zooplankton species (Weatherley 1988). However, the improvement of fish with liming and the associated abundance of zooplanktivorous fry, can reduce the numbers and mean sizes of copepods and cladocerans, which in turn ameliorate competition interactions and often result in an increase in rotifer populations (Weatherley 1988).

5.6. Liming and macroinvertebrates

Benthic macroinvertebrates include detritivores, grazers, and predators of invertebrates and fish. They are useful ecological indicators and are an important prey for many fish (Mant *et al.* 2011). Lake liming has been reported in some instances to cause an initial decline in abundance of benthic invertebrate populations and in others an increase in abundances of acid sensitive taxa present at the time of liming (Weatherley 1988; Hultberg & Andersson, 1982). Yet, the general responses of most invertebrate taxa have fallen short of expectations (Mant & Pullis 2012; Mant *et al.* 2011). When observed, positive population responses to liming have been attributed to reduce physiological stress, to improved food supplies or altered interspecific interactions (Weatherley 1988). Further, several macroinvertebrate taxa are known to be sensitive to acidity related conditions through effects on ion regulation and possibly respiration which liming may mitigate (Weatherley 1988). The increase in primary and secondary production and organic matter breakdown associated with liming is likely to provide an increase in food supply to most invertebrates (Weatherley 1988).

There are also certain aspects of the neutralise water chemistry that may restrict the establishment of invertebrates. For instance, acid run-off into the littoral zone of associated lakes may adversely affect some species, like *Astacus astacus* (Weatherley 1988). Profundal benthos taxa such as chironomids and oligochaetes, may also show a decline response after liming (e.g. Eriksson *et al.*, 1983). This may be associated with reductions in the sedimentation of organic matter from the epilimnion, or to the accumulation of precipitated Al compounds in this zone (Weatherley 1988; Eriksson *et al.*, 1983). Liming can also impact on streams, by depositing $CaCO_3$ locally on the stream benthos, or toxic conditions in mixing zones between limed and acid waters (Hesthagen & Larsen 2003). Other factors other than acid-base chemistry that can potentially limit invertebrate species responses include climate change, natural population variations, predation, dispersal abilities of acid-sensitive taxa and habitat factors such as food quality or structural physiography (Mant & Pullis 2012; Weatherley 1988).

5.7. Liming and fish

For mainly commercial reasons, the effects of acidification and liming on fish have received more attention than for other biotic components. In the majority of cases direct lake liming has increased the survival of acid sensitive species (Mant & Pullis 2012; Mant *et al.* 2011). Several investigations of liming in freshwater bodies have reported increased survival for a number of fish species including *Salmo trutta*, *S. gairdneri*, *Salvelinus salvelinus*, *S. alpinus, S. fontinalis*, *S. namaycush*, *Micropterus dolomieui*, *Percafluviatilis* and *Rutilus rutilus* (Mant & Pullis 2012; Mant *et al.* 2011; Clair & Hindar 2005; Weatherley 1988).

In running waters, liming of headwater lakes has resulted in increased juvenile densities of resident *S. trutta* and migratory *S. trutta* and *S. salar* (Weatherley 1988, Mant *et al.* 2011). Lime dosing in the River Vaaraana, Norway, greatly reduced the mortalities of caged *S. trutta* and *S. salar*, with no indication of a toxic zone immediately downstream of the additions (Weatherley 1988). Similarly, direct liming of a stream in Wales virtually eliminated mortalities of caged *S. trutta* and *Cottus gobio*, and reduced the mortality rate of *S. salar* (Weatherley & Ormerod 1992). However, extensive treatment throughout the 476 km^2 River Høgvads in catchment could not prevent *S. salar* mortalities associated with acid episodes in autumn (Edman & Fleischer 1980).

Fish kills in acidic waters are related to high Al concentrations, which cause stress in ion regulation and ventilation, particularly for pH levels of around 5 (Clair & Hindar 2005). In the few cases where liming has caused a fish kill, explanations have been attributed to a metal hydroxide precipitation onto gills and death due to combined osmoregulatory and ventilatory stress (Weatherley 1988). Such metal precipitations are likely to occur if metal concentrations were still high when the pH was elevated (Weatherley 1988). Flushing out of accumulated Al and Fe hydroxides from instream limestone gravel have also been recorded to cause catastrophic fish kills (Weatherley 1988). Nonetheless, avoidance behaviour from fish suggests that mobile life stages may minimize such risks (Weatherley 1988). Alkaline pH's may also be toxic through increased concentrations of hydrated metal ions, notably $Al(OH)_4$ (Weatherley 1988). Temporary accumulation of mobilised Hg by fish has been also observed in some instances in Sweden. In general however, Hg levels in fish tend to decrease after liming (Weatherley 1988).

In lakes with residual fish stocks where recruitment no longer exists, or is sporadic, liming invariably restores reproduction rapidly (Weatherley 1988; Eriksson *et al.*, 1983). This is due to enhanced survival of early life stages, especially larvae, which are often the most sensitive stage to acidity related factors (Weatherley 1988; Eriksson *et al.*, 1983). For instance, the incubation of eggs of *S. gairdneri* and *S. salar* in boxes of limestone gravel successfully increased the pH of interstitial waters and enhanced egg and yolk sac fry survival (Weatherley 1988). Yet the emergent fry are usually killed on the transition to ambient lake water. Hence, the application of this method may not be reliable in chronically acidified lakes but might be useful for protecting eggs and sac fry against acid episodes, or where interstitial waters are more acidic than surface waters (Weatherley 1988). Changes in water temperature and transparency following liming may however induce negative effect on fish populations (Weatherley 1988). Many fish are primarily visual predators so they may be particularly restricted by altered light penetration (Weatherley 1988).

In spite of the success of several stocking efforts to override the effects of acidification (Mant & Pullis 2012; Mant *et al.* 2011; Hultberg and Andersson 1982), liming has proved a necessary interim measure to preserve the genetic diversity of fish stocks (Clair and Hindar 2005; Weatherley 1988). Many cases of genetically distinct populations amongst salmonids, especially in Scandinavia, have been recognised, which are reflected in differences in vital phenotypic characters such as growth and spawning season (Weatherley 1988). Consequently, the local extinction of certain populations could therefore result on the loss of important cost-effective genetic traits. Equally important, there is also a potential danger of hybridisation between stocked fish with natural populations, which could lead to a loss of fitness by genetic introgression (the disruption of genotypes due to interbreeding) (Weatherley 1988). The recovery of different fish species will also be differentially affected by the various rates of recovery of other biotic components (Weatherley 1988; Mant *et al.* 2011). Thus acidification-induced perturbation followed by recovery could result in alternative equilibria (Weatherley 1988).

6. Liming in the UK

In comparison with Scandinavian countries, mitigation programmes based on lime application to surface waters and catchments in the UK have been limited and predominantly restricted to Wales and Scotland (e.g. Howells & Brown 1992, Howells & Dalziel 1995, Edwards *et al*. 1990 Ormerod & Durance 2009). Liming projects have been undertaken to improve conditions in acid waters, with applications to lakes, their tributaries or catchments (Howells & Brown 1992). In Cumbria for example, in response to the kill of spawning fish seen in the lower river during the springs of 1981 and 1984, lands adjacent to the river Esk were limed in 1986-87 with 3200 tonnes of powdered limestone. A decrease in the calcium concentration of the river water downstream was reported, but with pH values higher than before (Howells & Brown 1992). An increase in the run of spawning salmonids in the adjacent river Esk and even in the stream above the limed areas, were reported with no observed repetition of fish kill during the spring. Yet observations were difficult to attribute changes to liming (Howells & Brown 1992).

In several small catchments at Lyn Brianne on the Tywi in west Wales, restricted riparian liming within an afforested area (at 30 t ha^{-1}) was of limited value in improving water quality (Edwards & Stoner 2003). In another area, upland land management (draining, liming, ploughing) improved grazing productivity but had little effect on the chemistry of the stream draining to the treated area. In another treatment, however, application of limestone to a wetland source area (at 10 t ha^{-1}) led to rapid rise in the pH of stream water, although this improved water quality has not since been maintained (Edwards *et al*. 1990). After 25 years of monitoring this study showed that liming had few long-term benefits compared with natural recovery (Ormerod & Durance 2009, Durance & Ormerod 2007), and, according to Clair & Hindar (2005) this should be a key, general criterion in evaluating the outcomes of ecological restoration.

Another catchment liming study was conducted in Wales in the late 1980s on a 0.34 km^2 poorly buffered coniferous-moorland catchment that was limed with 9 t ha^{-1} of $CaCO_3$. The wetland portions of two other catchments were also limed using 20 and 25 t ha^{-1} each (Weatherley & Ormerod 1992). Results showed that plant, invertebrate, and fish communities partly recovered to pre-liming conditions in drainage streams. Acidophilic macrophytes disappeared from drainage streams before liming and did not appear after two years of improved conditions. Benthic invertebrate communities on the other hand showed a partial return to pre-acidic conditions. Despite the improvements in water quality being still notable 10 years after catchment liming, responses in the invertebrate community were modest (Bradley & Ormerod 2002). There were slight improvements in trout densities in the treatment streams, however, but these were also matched by improvements in the control sites. These matched responses of fish make any assertion inconclusive over the 2.5 years of follow-up study (Weatherley & Ormerod 1992). Considering the types of results reported for lakes and streams, it is most likely that recolonisation was affected by biotic factors not taken into account by the model, such as changes in the competitive environment and the “hit or miss” nature of recolonisation (Clair & Hindar 2005).

Another aspect of this liming study was reported by Waters *et al.* (1991), who used different liming strategies in two moorland catchments. They added 9 t ha^{-1} of limestone in a 33 ha area of one complete catchment. In the other, they applied 16 t ha^{-1} only in the hydrological source area, immediately surrounding the stream, covering 11% of the whole catchment area (Clair & Hindar 2005). Their results showed pH and Ca^{2+} increases, and Al decreases, at both sites after liming. The collected data also demonstrated that it was only necessary to lime hydrological source areas in order to improve the stream chemistry. They concluded that this more highly targeted approach was a much more efficient and cost-effective method in remediating stream chemistry, compared to a whole catchment liming (Clair & Hindar 2005). Elsewhere in west Wales a number of stocked lakes have been treated with direct application of limestone. This has improved water quality and the value of the “put-and-take” fisheries of brown trout there, although on time scales determined by time of water replacement (Underwood *et al.* 1987).

In the acid sensitive areas of Galloway, south-west Scotland, two schemes have been undertaken at Loch Dee and Loch Fleet (Howells & Brown 1992). In Loch Dee liming of the riparian zone of the lower White Laggan tributary and directly to the stream led to the apparent “loss” of most of the applied limestone in the lake, mostly associated with the sediments discharged by the White Laggan (Howells & Brown 1992). In this lake, average turnover is only 40 days, so that the loss of calcium carbonate in solution in the outflow is also a significant factor. In the event, fishery (*S.trutta*) production, thought to be declining in 1980, seems to have recovered in this not very strongly acidified lake (pH = 5.6), possibly by improved fishery management.

The Loch Fleet project was set up in 1984 for an initial period of 5 years (Howells & Brown 1992). The site was underlain by poorly buffering granites, and had shallow organic, peaty soils with moorland-type vegetation (Howells & Brown 1992). The project proponents used between 5 and 30 t ha^{-1} of lime in dust, slurry, or pellets on a number of catchments. Their approach in deciding on dosage was based on laboratory experiments, with an element of trial and error in the field. Their main goal was to try to provide acceptable water chemistry (increase pH to 6, $Ca^{2+} > 100$ µequiv L^{-1}, and ANC between 10 and 50 µequiv L^{-1}) for the survival of all stages of salmonids, where the fishery for brown trout (*S. trutta*) had been lost.

The Loch Fleet liming project had complex application schemes as it involved the addition of a number of treatments to various sub-catchments draining into the loch (Howells & Brown 1992; Howells & Dalzeil 1995). However, samples from the outlet stream of the loch, which integrated all sub-catchment responses, showed a rapid positive increase in pH from 4.5 to above 6 and in Ca^{+2} from <50 to 150 µequiv L^{-1}. These levels of improvement in the water chemistry were subsequently recorded up to 1995 (Dalziel 1995). In addition to acidic deposition, acid pulses in the streams caused by sea-salt exchanges also affected Loch Fleet. In response, Dalziel *et al.* (1994) indicated that catchment liming was the best method to alleviate these types of episodes in the Loch

Fleet streams. Based on the applied dosages on the catchments over the duration of the project, the authors predicted that improvements would last for another 9 years (Dalziel *et al.* 1994; Dalziel 1995).

The improvements in water chemistry in Loch Fleet and its inlet streams provide suitable conditions for survival of the various trout life stages within a few months of liming (Howells & Brown 1992). The lake was restocked with brown trout, which were found to be a successfully viable population 9 years after the liming treatment (Dalziel 1995). Alongside the improvement of fish populations, other biological groups also showed a major response including the occurrence of a number of alkalophilous periphytic and planktonic diatom species, some of which had never been found previously (Battarbee *et al.* 1992). Algal primary productivity also doubled after liming but no major long-term changes in benthic invertebrate populations in the loch were found. The authors suggested that the introduction of trout as top predators could potentially mask the recovery of invertebrates.

In addition to the above described response in the aquatic fauna, catchment liming in Loch Fleet showed also a number of moss species thriving or unaffected, while two acidophilic mosses, *Sphagnum capillifolium* and *S. papillosum*, were completely eradicated as soon as they came in contact with limed conditions (Howells & Brown 1992). Results in agreement with liming studies in Wales were that the liverwort *Nardia compressa*, was recorded to be greatly reduced in coverage after liming, and that no other species had moved in to replace it five years later (Wilkinson & Ormerod 1994). Gunn *et al.* (2001) described similar trends in the Sudbury, Ontario region for 15 wetlands limed with fine-particled dolomite. Other wetland plant communities showed negative effects at the same time as showing increases in pH.

Another liming project in the Kelty Water in central Scotland was reported by Miller *et al.* (1995). The area is part of the Loch Ard forest and is extensively stocked with conifers planted in the 1940s and 1950s, with some higher altitude planting in the 1980s. About 60% is forested, mainly with Sitka spruce (*Picea sitchensis*), with smaller areas of Norway spruce (*Picea abies*) and Lodgepole pine (*Pinus contorta*). Salmon (*Salmo salar)* are found in the lower, richer stretches of the system but are prevented from reaching the headwaters by a waterfall. Above this, a limited number of brown trout (*Salmo trutta*) existed and had access to a further 3 km of stream before encountering another waterfall. No salmonids are found upstream of these upper falls, i.e. within the study area, and the only fish encountered were eels (*Anguilla anguilla*). During the mid 1980s the hydrochemistry was not capable of supporting native fish populations with calcium concentrations around 20 µeq l^{-1}, less than the suggested critical value of 50 µeq l^{-1} and hydrogen ion concentrations around 70 µeq l^{-1}, greater than the critical value of about 30 µeq l^{-1}. In 1990 limestone was applied aerially to the source areas of selected streams, with around 5% (15 ha) of the total catchment area of 270 ha treated at 10 t ha^{-1}. Stream monitoring, carried out over the period 1989–1995, showed an immediate response to liming followed by a progressive decline. Calcium values were elevated to >150 µeq l^{-1} and hydrogen ion concentrations reduced to 20 µeq l^{-1}, reverting in time towards pre-liming values. Although salmonid survival was improved during low flow conditions in summer, only a few fry survived to the autumn as acid episodes increased, and these were subsequently lost from the system during the winter period. Budget calculations indicated losses of around 30% of the applied calcium during the first four years. Studies on the vegetation and soils revealed a greater than expected penetration of calcium to depth (10–20 cm) in the soil profile. Results suggest that source area liming at this rate has had minimal effects on the vegetation and by increasing the proportion of the catchment limed to 15% could have a much greater success in reducing the frequency of biologically damaging episodes (Miller *et al.* 1995).

7. Summary

In this report we have presented a general overview of the most common liming methods with their pros and cons as well as some potential effects of liming on aquatic biological communities. If a new liming project is being considered, it is of great importance to ensure that all proponents and regulators understand and address carefully the above described pros and cons and follow a clear process to ensure no further damage to the target ecosystems (Clair & Hindar 2005). It is equally relevant to address all possible negative and positive effects of liming and to have a reasonable chance of meeting the desired objectives (Clair & Hindar 2005). In order to help in achieving these two conditions and based on Clair & Hindar (2005), here we present a brief overview of the following issues that need to be addressed before any serious consideration of liming.

1. **Justification:** There has to be a good rationale to justify attempting to modify an ecosystem, and the parts of the ecosystem in need of protection need to be clearly identified. It is important to insure that even if a fish species is clearly in need of protection, other parts of the ecosystem (e.g. prey, predators and other competitors, access to habitat, other types of pollution effects) will not provide another obstacle to population recovery. Neglecting this step can only lead to the failure of a project.

2. **Biological and ecological understanding:** There must be clear understanding of the target species life cycle. If a fish species is to be recovered or protected, are all the vulnerable life stages protected? For example, there is no point improving a species' passage by liming a stream if the conditions at the spawning grounds are not allowing successful hatching to occur.

3. **Knowledge of the targeted area:** The proponents must have reasonable expectations of what is achievable with the methods they will be using. As the Norwegian experience shows, in order to devise a successful program on a large river system, there is often a need for more than one doser. Half measures are usually a waste of time in this kind of work.

4. **Understanding of recovery timing:** The proponents must have a good understanding of the time to recovery of the system. Dynamic acidification modeling reveals that water chemistry recovery may take decades in certain areas with poor

soils, such as southern Norway or Nova Scotia. There is no point beginning a liming program if proponents are only able to carry it out for a few years. This is clearly one of the major advantages to catchment liming as its effects can last for decades instead of hours in the case of in-stream dosing.

5. **Uncertainties:** often there are a number of other uncertainties associated with mitigation. Factors such as climate change with more extreme weather and reforestation can modify hydrological and pedological cycles in ways that might offset gains made by liming. These types of pre-project assessments need to be done as a first step in deciding on the suitability and desirability of mitigation approaches and will provide different results based on local conditions.

6. **Long-term feasibility:** ecosystem liming must be viewed as a tool to keep ecosystems or targeted fish populations from being irretrievably lost until nature can restore itself under less polluted conditions. It cannot be a substitute for pollution prevention, nor should it be used to create conditions that did not exist before the acidification problem existed. An important issue with liming is that it will likely need to be done over a long period of time in highly acidified systems, as re-acidification will occur once the effect is allowed to lapse. If this happens, any “positive” change in populations will be reversed, thus making the whole previous exercise a waste of time and money.

7. **Monitoring:** the effectiveness of liming on the target species and on wider ecosystem effects should be assessed by careful chemical and biological monitoring, ideally including before and after observations and the use of non-limed control streams with otherwise similar hydrology and chemistry, thus utilising the so-called BACI design (Before After Control Intervention).

8. References

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