NRC Research Press

REVIEW

Ecological benefits and risks arising from liming sugar maple dominated forests in northeastern North America

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Abstract: Liming, the application of carbonate materials (e.g., $CaCO_3$, $CaMg(CO_3)_2$) to soils and surface waters, has been used extensively in Europe, and to a lesser extent in Canada and the United States, to mitigate the effects of acid deposition on forest and aquatic ecosystems. This literature review was conducted to assess the effects of liming on ecologically and economically important sugar maple dominated ecosystems of northeastern North America, where it is increasingly used to treat sugar maple dieback. Potential direct and indirect effects were considered to determine whether the use of liming to revitalize these forests could negatively affect other ecological parameters, including those in adjacent aquatic habitats. Based on current scientific literature, it is not anticipated that liming at rates of 1–3 t ha^{-1} would have major detrimental effects on these ecosystems. However, liming could have negative effects on northern hardwood forests with regard to earthworm invasions. The choice of liming as a mitigation tool should be made not only after weighing the potentially negative effects against the benefits of restoring sugar maple dominated stands in poorly buffered soils, but also after considering ecological components that could be lost or never recovered if an acidified forest ecosystem is not limed.

Key words: acidification, acid rain, mitigation, liming, sugar maple, restoration, ecosystem, forest, aquatic habitat.

Résumé : Le chaulage, l'application de matériel carbonaté (par exemple: $CaCO_3$, $CaMg(CO_3)_2$) sur les sols et les eaux de surface, a été largement utilisé en Europe, et dans une moindre mesure au Canada et aux États-Unis, afin d'atténuer les effets des dépôts acides sur les écosystèmes forestiers et aquatiques. Cette revue de littérature a été effectuée afin d'évaluer les effets du chaulage sur les écosystèmes forestiers dominés par l'érable à sucre, qui sont importants au plan écologique et économique dans le nord-est de l'Amérique du Nord, dans lesquels il est de plus en plus utilisé pour traiter le dépérissement de l'érable à sucre. Les effets potentiels directs et indirects ont été considérés afin de déterminer si l'utilisation du chaulage pour revitaliser ces forêts pourrait avoir des effets négatifs sur d'autres composantes écologiques, incluant celles des habitats aquatiques situés à proximité. Sur la base de la littérature scientifique actuelle, il n'est pas prévu que le chaulage à des taux de 1 à 3 t ha^{-1} ait des effets néfastes importants sur ces écosystèmes. Cependant, les érablières chaulées pourraient être sujettes aux effets négatifs liés à l'invasion de vers de terre. La décision finale d'utiliser le chaulage comme outil de mitigation doit non seulement tenir compte des effets potentiellement positifs et négatifs de ce traitement dans les érablières situées sur des sols à faible pouvoir tampon, mais aussi des composantes écologiques qui pourraient être perdues ou ne jamais être récupérées si un écosystème forestier acidifié n'est pas chaulé.

Mots-clés : acidification, pluie acide, mitigation, chaulage, érable à sucre, restauration, forêt, habitat aquatique, écosystème.

1. Acid deposition and the need for liming

Acid deposition (acid rain) emerged as one of the most publicly visible environmental issues during the 1970s and 1980s (Driscoll et al. 2001). In acid-sensitive regions of North America, considerable effort was devoted to documenting the effects of acidification on forest, lake, and stream ecosystems. Studies showed that acid deposition had negative effects on soils (Houle et al. 1997; Likens et al. 1998; Ouimet et al. 2001; Bailey et al. 2005; Long et al. 2009), soil fauna (Kuperman et al. 2002), forest tree species (DeHayes et al. 1999; Ouimet et al. 2001; Sullivan et al. 2013), amphibians (Pough 1976; Pough and Wilson 1977; Dale et al. 1985; Freda 1986; Pierce 1993; Brodkin et al. 2003), birds (Scheuhammer 1991; Hames et al. 2002; Pabian and Brittingham 2011), aquatic biota (Baker et al. 1993; Vinebrook et al. 2002; Walseng et al. 2003; Nierzwicki-Bauer et al. 2010; Lacoul et al. 2011), mammals (Scheuhammer 1991), and even humans (Héroux and Jessiman 2005).

Efforts to mitigate these effects brought about reductions in the emissions of precursors of acidic deposition (i.e., sulphur (SO_x) and nitrous (NO_x) oxides). The Convention on Long-range Transboundary Air Pollution in Europe in the 1980s, the United States – Canada Air Quality Agreement in the 1990s, and the Title IV of the 1990 Clean Air Act Amendments in the United States subsequently led to major reductions in SO_2 emissions. In North America, reductions in emissions resulted in both corresponding decreases in sulphur deposition to terrestrial ecosystems and declining sulphur concentrations in surface waters (cf. Driscoll et al. 1996). Similar regulations have also targeted atmospheric nitrogen deposition in North America (US EPA 2010) and have resulted in significant reductions in NO_x emissions and deposition, both in the United States (US EPA 2010; Pinder et al. 2011; Templer et al. 2012) and in Canada (Geddes et al. 2009). Unlike NO_x emissions, NH_3 emissions are unlikely to decline, and an increasing trend has

Received 4 June 2014. Accepted 17 November 2014.

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been reported in the United States (Templer et al. 2012). Overall, global atmospheric nitrogen deposition has doubled in the past century and is projected to more than double in the next, mainly due to fossil fuel combustion and growing demand for nitrogen by agriculture and industry (Galloway et al. 2004, 2008). As a result, rates of acidic deposition still exceed soil neutralizing capacity in many regions (Ouimet et al. 2006; Duarte et al. 2013). The recovery of affected ecosystems has been slow (Nierzwicki-Bauer et al. 2010; Lawrence et al. 2012), and some areas may not recover unless further measures are undertaken (Sverdrup et al. 2005). Since the effects of acid deposition remain a concern (Menz and Seip 2004), there is continuing interest in the application of alkaline materials as a means of restoring acid-affected systems (Clair and Hindar 2005).

2. Liming of forest and aquatic ecosystems

Liming, the application of calcium-rich materials, is one of the oldest methods of improving agricultural soils, and has been used for hundreds of years. However, liming forest and aquatic ecosystems is a relatively new practice. To counteract the effect of acid rain and soil acidification on forest and aquatic ecosystems, liming has been carried out in Europe (Hultberg and Andersson 1982; Smallidge et al. 1993; Clair and Hindar 2005; Löfgren et al. 2009; Hesthagen et al. 2011) and in northeastern North America (Safford and Czapowskyj 1986; Auchmoody and Filip 1973; Smallidge et al. 1993; Wilmot et al. 1996; Clair and Hindar 2005; Ouimet et al. 2008; Moore and Ouimet 2010, 2014; Long et al. 2011; Moore et al. 2012; Pabian et al. 2012). In Europe, liming was used in the 1980s and 1990s to remedy the widespread crown thinning and dieback affecting European forests that was attributed to acid deposition (Huettl and Zoettl 1993). In Sweden (Henrikson and Brodin 1995) and Norway (Sandøy and Romundstad 1995; Hesthagen et al. 2011), liming is a common means to counteract surface water acidification. Since the 1970s, 7500 lakes, 11 000 km of watercourses, and 1600 mires have been limed in Sweden (Guhrén et al. 2007). In Germany, nearly one third of the forested area has been limed since the 1980s (Anonymous 2003). Liming is still used in parts of Germany to protect groundwater from metals such as aluminum and cadmium and to enhance seedling regeneration (Hairsine 2014).

In northeastern North America, the use of lime in forest ecosystems and adjacent aquatic habitats started modestly in the 1940s (Cummings 1941) and in the 1950s to restore pH of acid lakes and streams (Kretser and Colquhoun 1984). Although liming of aquatic habitats is still practiced, liming is mainly used today to restore the vitality of sugar maple (*Acer saccharum* Marsh.), one of the most abundant hardwoods and an economical and aesthetically important species in forests of eastern Canada and the northeastern United States (Godman et al. 1990). Forest or watershed liming can also provide long-term neutralization of surface water. Although this technique is not yet widely used in operational settings, forest liming has received growing interest due to the need for mitigation tools to counteract the decreasing vitality and productivity of sugar maple dominated stands observed in many areas of northeastern North America over the last decades (Wilmot et al. 1996; Horsley et al. 2002; Hallett et al. 2006; Bailey et al. 2004; Schaberg et al. 2006; Gavin et al. 2008; Tominaga et al. 2008; Long et al. 2009, 2011; Watmough 2010; Moore et al. 2012). This has also provided a better understanding of the role of calcium and pH in these ecosystems.

Several studies have documented the beneficial effects of liming on declining sugar maple dominated forests (Wilmot et al. 1996; Moore and Ouimet 2010; Long et al. 2011; Moore et al. 2012). Liming has been used recently to treat declining sugar maple dominated stands, particularly in Quebec. A survey among mechanical spreader manufacturers and spreading companies (Fig. 1) showed that about 10 000 ha of forests have been limed over the last 5 years in Quebec. It seems, however, that liming is not yet a standard practice elsewhere in the northeastern United States (T.D. Perkins, University of Vermont, pers. comm.). Also, the use of lime has been reported in the last decades to treat watersheds that have been affected by acidic deposition, acid mine drainage, or both (Clayton and Menendez 1996; Gunn et al. 2001; Turner et al. 2003; Sharpe et al. 2006; DEC 2014).

Fig. 1. Example of a mechanical spreader used in sugar maple dominated forests of Quebec (photo from Danago Inc.).

Though many efforts have been made to review the available studies documenting the effect of liming on aquatic ecosystems (Weatherley 1988; Porcella et al. 1990; Howells and Dalziel 1992; Henrikson and Brodin 1995; Clair and Hindar 2005), reviews focusing on terrestrial ecosystems are rare (Smallidge et al. 1993). The objective of this paper is to review the current state of knowledge on liming effects to assess potential impacts on sugar maple dominated forests and associated ecosystem components such as aquatic habitats. Knowing and understanding the effects of liming on acidic, base-poor sugar maple dominated stands is important to determine the ecological suitability of this practice in the future.

3. Types of liming materials and application methods

Limestone and dolostone, two sedimentary carbonate rocks, as well as marble and dolomitic marble and their metamorphic equivalents, are all mined, ground, and used in various regions as liming materials for aquatic (Weatherley 1988; Henrikson et al. 1995) and terrestrial applications (Smallidge et al. 1993; Moore et al. 2012) in acidified ecosystems. This is mainly due to their low cost, easy handling, and low contaminant content (Henrikson et al. 1995). Their magnesium content can vary greatly, with calcite ($CaCO_3$) having the lowest magnesium content (0–6%), and dolomite ($CaMg(CO_3)_2$) having the highest (magnesium ≥ 12%). Additional lime chemical types include burnt lime (CaO) and hydrated lime ($Ca(OH)_2$), but they should be avoided because of their high reactivity that can cause foliage burn, among other phenomena, and which poses handling risks. Other materials sometimes used for calcium addition include gypsum ($CaSO_4{\cdot}2H_2O$), which increases calcium availability but generally does not affect soil pH (Shainberg et al. 1989; Bigelow and Canham 2007) in the same way as wollastonite (CaSiO3; Groffman et al. 2006) or limestone. Fertilizing residue materials with some liming power such as wood ash, marl, shells, and certain industrial by-products such as paper-mill lime-sludge and concrete dust have been used only experimentally in northern hardwood forests (e.g., Feldkirchner et al. 2003).

Lime has been applied by helicopter over the forests in Germany since the mid-1980s to counter forest soil acidification (Huettl and Zoettl 1993). In North America, calcium materials have been applied by helicopter both for watershed experiments (e.g., Driscoll et al. 1996; Battles et al. 2014) and for lake and stream remediation (Hudy et al. 2000; DEC 2014) in the northeastern United States. Lime has been applied by hand, mechanical spreader, or airplane to counter sugar maple decline in Quebec, Canada (Côté and Ouimet 1996). In the early 2000s, the creation of wheeled mechanical spreaders pulled by tractors or all-terrain vehicles has greatly facilitated the application of lime and fertilizer materials in sugar maple stands, particularly in Quebec (Fig. 1). More than 100 spreaders have been sold to maple syrup producers over the last 6 years in this province (Sylmar equipments Inc.; Conception Duquette Inc.).

Several methods have been proposed to estimate the amount of lime required for soils, but ideally this should be done in connection with nutritional needs for plant vitality and growth (Smallidge et al. 1993). For sugar maple, many studies have been conducted to better establish the nutritional needs of this species depending on site fertility (Lozano and Huynh 1989; see Moore and Ouimet 2010 for a review; Ouimet et al. 2013). Based on preliminary results over a 15 year period, a liming rate of approximately 3 t ha^{-1} may be sufficient to restore sugar maple nutritional status, depending on the initial soil nutrient status (Moore et al. 2012). Liming is sometimes combined with other fertilizers and, therefore, isolating these factors can be difficult when assessing the effects of liming on ecosystems.

4. Liming effects on forest ecosystems

Table 1 summarizes the effects of calcium addition on organisms inhabiting sugar maple dominated ecosystems (forests and lakes) of northeastern North America.

4.1. Forest stand and catchment

Over the last decades, evidence of base cation depletion in soils with a low acid-buffering capacity has been reported in forests of northeastern North America (Houle et al. 1997; Likens et al. 1998; Johnson et al. 2008). Many studies have suggested that this phenomenon was caused, at least in part, by acid deposition (Houle et al. 1997; Likens et al. 1998; McLaughlin 1998; Sharpe 2002; Bailey et al. 2005; Long et al. 2009). Sugar maple is known to be very sensitive to soil acidity, and poor soil calcium content (Wilmot et al. 1995; Bailey et al. 2004; Long et al. 2011; Moore et al. 2012; Sullivan et al. 2013). Duchesne et al. (2002) showed that the appearance of the sugar maple decline syndrome and associated growth reduction in Quebec can be related, at least in part, to soil acidification and acid deposition levels. In this context, it is not surprising that many studies on sugar maple suggested that base cation deficiency, and particularly calcium and magnesium deficiencies, were a cause of the growth reduction, dieback, and lack of regeneration of trees (Wilmot et al. 1995; Bailey et al. 2004; Schaberg et al. 2006; Huggett et al. 2007; Long et al. 2009; Watmough 2010; Duchesne et al. 2013; Sullivan et al. 2013; Battles et al. 2014). Huggett et al. (2007) suggested that the cumulative depletion of calcium caused important reductions in crown vigor, radial growth, and wound closure of sugar maple trees.

Moreover, studies in acidic soils of northern Pennsylvania revealed that high levels of foliar manganese (>2.0 mg g^{-1}) existed in mature sugar maples (Horsley et al. 2000; Kogelmann and Sharpe 2006). McQuattie and Schier (2000) observed manganese toxicity for container-grown sugar maples. In this context, Kogelmann and Sharpe (2006) hypothesized that the manganese could be associated with poor sugar maple health on soils vulnerable to acidification. However, no symptoms of manganese toxicity in field-grown trees have been observed for sugar maple or any other northeastern species (Long et al. 1997).

Given that the soil conditions in many sugar maple dominated stands no longer seemed able to maintain sugar maple, liming was proposed as a tool to restore conditions that existed prior to soil acidification. Liming usually targets declining and base-poor sugar maple dominated stands located on private lands. Although some streams and ponds can receive lime material when a forest is limed, lakes and rivers are normally not directly exposed to spreading. This "strategic catchment liming" (Farmer 1992) avoids potentially detrimental practices, such as direct application of lime to wetland habitats. Studies conducted in sugar maple dominated stands have shown that liming may enhance sustainability by promoting long-term growth, vigor, and regeneration of sugar maple (Wilmot et al. 1996; Long et al. 2011; Moore et al. 2012), and that these stands do not require frequent treatments. Moreover, Moore et al. (2012) suggested that low lime doses, in the range of 3 t ha^{-1}, may be sufficient to revitalize sugar maple trees over the long term. In their study, the basal area increment of limed sugar maple trees more than doubled (+114%), and crown dieback dropped from 39% to less than 10% 15 years after application of 2 and 5 t ha^{-1}, as compared to the control (unlimed) trees. However, the effect of calcium addition on sap quality and yield has not yet been formally demonstrated. At the Hubbard Brook Experimental Forest in New Hampshire, calcium amendment with silicate material improved photosynthetic surface area, aboveground net primary production, tree biomass increment, reproductive ability, and extension growth (for seedlings) of sugar maple trees in the treated watershed, compared to that in the reference watershed (Halman et al. 2013, 2014; Battles et al. 2014; Marlow and Peart 2014). Another study showed that calcium addition facilitated sugar maple wound closure and injury recovery from ice storm damage (Huggett et al. 2007). Although some studies have suggested that sugar maple composed a much smaller portion of the presettlement forest than of the current forests of northeastern North America (e.g., Cogbill et al. 2002), other reports suggest that sugar maple has always been among the dominant tree species in the areas studied (e.g., Fuller et al. 1998; Dyer 2001). These contrasting results could be due to differences in both site characteristics and in historical natural and anthropic disturbances of the studied areas. One way or another, liming seems warranted as a restoration or management tool for this aesthetically, biologically, and economically important species in contemporary forests.

In contrast to sugar maple, the addition of 22 t ha^{-1} of $CaMg(CO_3)_2$ had a negative effect on black cherry (*Prunus serotina*) growth and survival in Pennsylvania (Long et al. 2011), and no effect on American Beech (*Fagus grandifolia*) (Long et al. 2011). In Quebec, the application of 3 t ha^{-1} of $CaCO_3$ had also no effect on beech (Duchesne et al. 2013). More studies are needed to determine the optimal liming rate to restore sugar maple in a given area without affecting other valuable species.

Sullivan et al. (2013) have linked the near absence of sugar maple seedling regeneration in sugar maple dominated stands in the Adirondacks (New York), receiving high levels of atmospheric acid deposition, to a low exchangeable calcium and low percent base saturation. In another study, a pelletized wollastonite ($CaSiO_3$) addition at a rate of 0.85 Mg Ca ha^{-1} to a watershed at Hubbard Brook Experimental Forest was used to replace calcium depleted from soils. This treatment resulted in a significant increase of sugar maple seedling density and survival (Juice et al. 2006). However, more recent research has shown that a decrease of calcium availability over time reduced the positive responses of sugar maple seedlings to this treatment (Cleavitt et al. 2011).

Not all tree species are declining in acidified sugar maple dominated stands. For example, American beech seems unaffected by the current acidity of forest soils, and black cherry growth is better on very acid sites (Long et al. 1997, 2009; Duchesne et al. 2005, 2013; Duchesne and Ouimet 2009; Sullivan et al. 2013). Studies indicate that forest dynamics are already shifting toward a domination of beech in some ecosystems of northeastern North America (Jenkins et al. 1999; Duchesne et al. 2005; Gravel et al. 2011; Sullivan et al. 2013). This could be problematic, both from an ecological and an economical viewpoint. From an ecological

Table 1. Effects of Ca addition in forests and adjacent aquatic habitats of northeastern North America.

	Effect		
Organisms	Positive	Negative	None or mixed
(*a*) Forest ecosystems*			
Tree and regeneration			
American Beech			Long et al. 2011; Duchesne et al. 2013; Halman et al. 2014; Marlow and Peart 2014
Black cherry		Long et al. 2011	
Sugar maple	Wilmot et al. 1996; Juice et al. 2006; Huggett et al. 2007; Long et al. 2011; Moore et al. 2012; Halman et al. 2013; Battles et al. 2014; Halman et al. 2014; Marlow and Peart 2014		
Other plants	Brach and Raynal 1992 (oxalis); Demchik and Sharpe 2001 (grass, violet, *Carex*); Pabian et al. 2012 (forb)	Brach and Raynal 1992 (*Lycopodium*); Gunn et al. 2001 (*Polytichum*)	Demchik and Sharpe 2001 (ginseng); Pabian et al. 2012 (grass, fern, shrub)
Amphibian			
Red-backed salamander			Seagle and Curd 1994; Moore 2014
Bird	Pabian and Brittingham 2007; Pabian et al. 2012		
Soil flora and fauna			
Armillaria			Marçais and Wargo 2000
Mycorrhiza	Coughlan et al. 2000; St. Clair and Lynch 2005; Juice et al. 2006		
Microarthropods		Chagnon et al. 2001; Fisk et al. 2006	Fisk et al. 2006
Macroarthropods			
Earthworm	Bernard et al. 2009; Moore et al. 2013		
Millipeds		McCay et al. 2013	Pabian et al. 2012
Slug			Skeldon et al. 2007
Snail	Pabian and Brittingham 2007; Skeldon et al. 2007; Pabian et al. 2012; McCay et al. 2013		
Spider		McCay et al. 2013	Dow 2011
(*b*) Adjacent aquatic habitats†,‡			
Fish	Popp et al. 1996; Clayton et al. 1998; Hudy et al. 2000; McClurg et al. 2007		LeFevre and Sharpe 2002
Trout	Schofield and Keleher 1996; Clayton et al. 1998; Hudy et al. 2000; McClurg et al. 2007		Eggleton et al. 1996
Vegetation			
Algae	Bukaveckas 1989	Molot et al. 1990	Roberts and Boylen 1989
Macrophyte	Weiher et al. 1994		Hagley et al. 1996
Invertebrate	Clayton and Menendez 1996		LeFevre and Sharpe 2002
Benthic	Hudy et al. 2000; McClurg et al. 2007		Eggleton et al. 1996
Zooplancton	Schaffner 1989		Wright et al. 1996

*Also see Smallidge et al. (1993) for a review.
†Also see Clair and Hindar (2005) for a review.
‡Also see Mant et al. (2013) for a review.

perspective, the shift in composition could lead to a loss of species associated to maple-dominated forests. Economically, this may be problematic because of the low value of beech compared with sugar maple. In these ecosystems, liming should therefore be used to preserve or restore sugar maple and forest biodiversity.

Some studies have shown that calcium addition could lead to unexpected outcomes in northern hardwood stands. Green et al. (2013) suggest that calcium addition increased annual evapotranspiration, causing a decrease in water flowing from a calcium-amended forest in New Hampshire. However, this increase was short-term (3-year), and seems to have been related to the return of sugar maple vigor to its pre-acidification level. Another outcome could have a more prolonged effect in the ecosystem. Because an increase in soil pH following liming may facilitate earthworm invasion and because earthworms may affect forest dynamics, sugar maple stand dynamics could be affected by earthworm colonization. In Minnesota, Larson et al. (2010) found that sugar maple trees growing in invaded conditions were more sensitive to drought than trees growing in earthworm-free conditions. Other studies suggest that worm selectivity in seedling ingestion and their alteration of seedbed conditions can profoundly change plant community composition in deciduous forests (Hale et al. 2005, 2006; Frelich et al. 2006; Holdsworth et al. 2007; Eisenhauer et al. 2009). The effects of liming on earthworms will be discussed further in section 4.3.

4.2. Soil chemistry

Long-term studies of sugar maple dominated stands in northeastern North America have shown that forest floor liming increased pH and base cation (e.g., calcium, magnesium) concentrations, and decreased exchangeable acidity, particularly in the upper soil horizons (Long et al. 1997; Moore et al. 2012). These changes were also reflected in foliage nutrient composition, with an increase in base cation concentrations (Long et al. 2011; Moore et al. 2012). A decrease in the concentration of potentially toxic elements (e.g., aluminium, manganese) has also been observed in soil and foliage (Long et al. 2011; Moore et al. 2012). High liming rates (10–50 t ha^{-1}) have been shown to decrease soil concentrations for some nutrients such as nitrogen, phosphorus, and potassium, and to have negative effects on sugar maple nutrition (Long et al. 2011; Moore et al. 2012). However, they did not affect tree growth and crown vitality. In contrast, low to moderate liming rates (from 0.5 to 5 t ha^{-1}) did not decrease concentrations of these elements in the soil or foliage (Moore et al. 2012; Pabian et al. 2012).

In a whole-watershed liming study in the northeastern United States, mass balance calculations showed that liming with ~7 t ha^{-1} of $CaCO_3$ increased the flux of dissolved organic carbon from soils to streams (Driscoll et al. 1996). Nineteen years later, the forest floor of this mixed hardwood forest had more than doubled the size of its carbon reservoir in response to the liming treatment (Melvin et al. 2013). In another experiment involving dolomitic lime rates ranging from 2 to 20 t ha^{-1}, no such change in forest floor carbon accumulation could be detected after 10 years (Moore et al. 2008). The results of these two long-term experiments indicate that, surprisingly, liming did not stimulate the decomposition rate of the organic matter in the forest floor of these northern hardwood forests. Greater amounts of dead organic matter through litter or root inputs do not appear to drive changes in the forest floor carbon dynamics in response to liming. In addition, the fact that forest floor and mineral soil carbon/nitrogen ratio showed no major change after liming (Moore et al. 2012; Melvin et al. 2013) indicates that organic matter decomposition is not stimulated by this treatment.

Nitrate leaching slightly increased in limed watershed (Driscoll et al. 1996). This suggests that liming favours soil nitrification. However, this acidifying process is largely buffered by the gain in acid neutralizing capacity caused by the treatment. In coniferous forests of Europe, studies have associated liming to greater nitrate concentrations in runoff, faster organic matter decomposition, and increased heavy metal mobilization (Kreutzer 1995). However, some authors suggested that nitrogen uptake by the vegetation could counteract leaching losses (Lundell et al. 2001). The ability of vegetation to counter leaching losses may depend in part on the timing of lime application in relation to the growing season, climatic conditions such as rainfall and soil temperature, as well as rates of atmospheric nitrogen deposition (Bäckman and Klemedtsson 2003; Trammell et al. 2004).

4.3. Soil fauna and flora

Changes in soil chemistry following lime addition can have direct effects on soil biota, especially on organisms sensitive to changes in pH and base cation content. However, it is difficult to distinguish whether the response of soil biota to liming is driven mainly by pH or base cations, since these soil properties are strongly interrelated. Of particular concern to forest managers are the effects of liming on soil organisms that are directly involved in organic matter cycling and nutrient availability.

The compilation of the effects of calcium addition on soil invertebrates shows a highly variable response of this group depending on the treated area; this suggests that the effects could be site-specific. Foster et al. (1995) argued that the effects of liming are much smaller than those associated with natural annual variations in assemblage composition, and far less than those associated with afforestation. Some studies performed in hardwood forests of northeastern North America reported a negative effect of calcium addition for some soil decomposer groups (2 and 20 t ha^{-1} $CaMg(CO_3)_2$, Chagnon et al. 2001; 1.6 t ha^{-1} $CaSiO_3$, Fisk et al. 2006; 10 t ha^{-1} $CaCO_3$, McCay et al. 2013), while others reported no effect (Fisk et al. 2006; 1.6 t ha^{-1} $CaSiO_3$, Skeldon et al. 2007; 4.5 t ha^{-1} $CaMg(CO_3)_2$, Pabian et al. 2012) or beneficial effects (4.5 t ha^{-1} $CaMg(CO_3)_2$, Pabian and Brittingham 2007; Skeldon et al. 2007; Pabian et al. 2012; McCay et al. 2013). In a laboratory study, Tompson et al. (2013) showed a beneficial effect of liming on isopods. Other invertebrates that do not directly feed on leaf litter, but which can regulate decomposer populations, have been found to be unaffected (10 t ha^{-1} $CaCO_3$, Dow 2011) or negatively affected (McCay et al. 2013) by calcium addition. European studies reported similar mixed results of negative (Hågvar and Kjøndal 1981; Hågvar and Amundsen 1981; Hågvar 1984; Ormerod and Rundle 1998; Haimi and Mätäsniemi 2002), no or positive effects (Adams et al. 1978; Gärdenfors 1992; Foster et al. 1995; Korenko et al. 2008) on soil invertebrates following calcium additions. In France, a smaller density of macroinvertebrates was observed in limed soil compared to the control, whereas community richness was unaffected (Auclerc et al. 2012). Smallidge et al. (1993) concluded that coniferous stand liming generally has a positive effect on microbial populations. In a laboratory study, Persson et al. (1989) found a negative effect of liming a coniferous mor humus on soil/root nematodes, particularly on the bacterial feeders and omnivores, but not on the obligate predators.

While mycorrhization of sugar maple roots may be reduced at low soil pH (Ouimet et al. 1995, Coughlan et al. 2000; St. Clair and Lynch 2005), it can be stimulated by calcium addition (Coughlan et al. 2000; St. Clair and Lynch 2005; Juice et al. 2006). This symbiosis is important to the inorganic nutrition of host plants (Smith and Read 1997) and could enhance plant growth and survival at low soil pH (Clark 1997). However, a short-term laboratory study did not find any effect of mycorrhization on seedling nutrition and growth of sugar maple (Ouimet et al. 1996). Marçais and Wargo (2000) showed that liming increased the abundance of *Armillaria* rhizomorphs in treated plots; however, rhizomorph frequency on sugar maple root collars was not greater in lime-treated plots than in control plots. Observations made in the 1990s indicated that *Armillaria* root disease was widespread across the Allegheny Plateau in northern Pennsylvania, and that the fungus was causing mortality in declining sugar maple dominated stands of this area and in surrounding state forests (P.M. Wargo, USDA Forest Service, unpublished results).

The lack of response of foliar nitrogen and phosphorus concentrations in sugar maple trees, 15 (Moore et al. 2012) and 22 years (Long et al. 2011) following calcium addition, suggests that liming has no long-term effect on soil decomposers as a whole in sugar maple dominated stands. In contrast, Melvin et al. (2013) suggest that the increase of forest floor carbon and nitrogen stocks in a mixed hardwood forest of the Woods Lake Watershed (New York), 20 years following calcium addition (6.9 t ha^{-1} $CaCO_3$), was due to reduced decomposition in the treated area. Other factors, however, might have limited the capacity of decomposers to respond to increased soil pH and calcium content (Groffman et al. 2006; Melvin et al. 2013).

Some of the detrimental effects observed in some studies may have been caused by excessive doses of lime. The observed negative effect of calcium addition on some invertebrates could also just be a return to equilibrium conditions that prevailed before soil acidification. For example, a decrease in abundance of some microarthropod species in acidic, base-poor northern hardwood soils was attributed to the loss of acidophilic species after calcium addition (Chagnon et al. 2001; Fisk et al. 2006).

Another important consideration regarding liming effects is the response of earthworms to this treatment, and their effects on soil and forest floor biota. Liming studies in northeastern North America (Bernard et al. 2009; Moore et al. 2013) and Europe

(Robinson et al. 1992; Ammer and Makeschin 1994; Kreutzer 1995; Potthoff et al. 2008) have documented increases in earthworm abundance. Earthworms generally prefer neutral soils, and their density and diversity are usually low in very acid soils (Ammer and Makeschin 1994; Curry 1998; Rusek and Marshall 2000; Chan and Mead 2003). The Enchytraeidae are acidophilic species and usually react negatively to liming, but the presence and abundance of basiphilic species generally increase after liming (Rusek and Marshall 2000). Of particular concern is that increases in soil pH following liming may facilitate earthworm colonization. Invasion by earthworms has been shown to have both direct and indirect effects on many components of the forest ecosystem (Gundale et al. 2005; Bohlen et al. 2004*a*, 2004*b*; Hale et al. 2006; Suárez et al. 2006; Eisenhauer et al. 2007; Maerz et al. 2009; Loss and Blair 2011) and its dynamics (Eisenhauer et al. 2009; Nuzzo et al. 2009; Larson et al. 2010). In Minnesota, invasion by earthworms in some areas has been associated with the extirpation of a rare fern species (Gundale 2002). In Minnesota and Wisconsin, Holdsworth et al. (2007) showed that earthworm invasions may be an important cause of reduced plant-species richness in forests dominated by sugar maple. In New York, Maerz et al. (2009) showed that the loss of leaf litter following invasion by non-native earthworms decreased the abundance of the eastern red-backed salamander (*Plethodon cinereus*), partly because of declines of small arthropods that are its food source. However, Ransom (2011) showed that the refuge provided by earthworm burrows increased survival of this salamander species in Virginia. This suggests that the effects of earthworm invasions may be region or habitat specific (Maerz et al. 2009).

It should be noted that native earthworms are rare in northeastern North America because they were eradicated during the last glacial period (Gates 1976). Non-native European species, such as *Lumbricus terrestris*, have been expanding their distributions since the European settlement (Gates 1982; Holdsworth et al. 2007). Invasion of Asian earthworms of the genus *Amynthas* is more recent in the northeastern United States than that of lumbricids (Bernard et al. 2009; Greiner et al. 2012). The many species involved in the invasions have different calcium and pH sensitivities, and they affect soils in various ways. Although some species may tolerate acid conditions (e.g., small epigeic earthworms such as *Dendroboena* and *Dendrodrilus* spp.), most previous reports of earthworm invasions associated to important changes in forest ecosystems have involved large acid-intolerant species, such as *Lumbricus terrestris* (see Moore et al. 2013).

4.4. Herbaceous and understory vegetation

Few studies have evaluated the effects of liming on ground vegetation in northern hardwood forests. Three years after liming, Pabian et al. (2012) observed an increase in forb ground cover, but no changes in other vegetation. Demchik and Sharpe (2001) reported that the abundance of some species increased significantly after liming, although overall ground plant diversity was not modified. Brach and Raynal (1992) reported a positive effect of liming on *Oxalis acetosella*, but a negative effect on the acidophil *Lycopodium lucidulum*.

In Europe, adverse effects of liming on vegetation are most commonly associated with wetland habitats, particularly because of mortality among species of *Sphagnum* (Mackenzie 1992; Wilkinson and Ormerod 1994; Bragg and Clymo 1995; Hindar et al. 1996; Larsson 1995; Traaen et al. 1997). Loss of *Sphagnum* may profoundly alter the structure and function of bog communities (Bragg and Clymo 1995). However, this undesirable effect is mainly observed when lime is applied to a wetland at an excessive rate or in finely ground form (Bragg and Clymo 1995; Henrikson et al. 1995). In the region of Sudbury (Ontario), Gunn et al. (2001) observed some mortality of *Polytrichum* in wetlands following liming at a high rate (32 t ha^{-1}), but not for other vegetation. In contrast, Mackun et al. (1994) showed that a low rate of lime addition (1.1 t ha^{-1} $CaCO_3$) had very few short-term impacts on wetland vegetation in the Adirondack Region of New York, and no negative impact on species of *Sphagnum*. Henrikson et al. (1995) suggested that more frequent applications of smaller doses of coarser liming material would reduce unwanted effects. As wetland habitats are not typically associated with sugar maple dominated forests, these effects are of lesser concern than potential impacts on forest understory communities.

Several studies report enhanced colonization of exotic, invasive plant species in less acidic, more fertile sites (e.g., Proulx and Mazumder 1998; Qaderi et al. 2009). Both fertilization and entry into forests with mechanized equipment, which may spread seeds and propagules, may cause liming to favor invasions. As invasive plants are a major management concern in many forests, this potential impact is at least worth mentioning as a topic for further study.

4.5. Terrestrial macrobiota

Only two studies have evaluated the direct effect of liming on herpetofauna in northeastern North America (Seagle and Curd 1994; Moore 2014). Both target the eastern red-backed salamander, which is well distributed (Moore and Ouellet 2014) and among the most abundant amphibians in northern hardwood forests (Petranka 1998). Both studies reported no short-term effect of liming on this species. Moreover, studies have shown that the red-backed salamander can survive in a wide range of pH (3–7.5: Wyman and Jancola 1992; Moore and Wyman 2010); this suggests that changes in soil pH should not affect this species in the longer term. In Europe, a positive effect of liming on the breeding success of some frog species was reported in the early 1990s (Bellemakers and van Dam 1992; Beattie and Tyler-Jones 1992; Beattie et al. 1993).

Some studies have suggested that the decline in calcium availability in forest ecosystems has negatively affected reproductive output of birds in Europe (Graveland and Drent 1997; Tilgar et al. 2002) and northeastern North America (Blum et al. 2001; Pabian and Brittingham 2007; Pabian et al. 2012) because of a reduction in the availability of calcium-rich preys, such as snails (Graveland et al. 1994), or in the calcium content of preys. Other studies have examined whether the restoration of calcium status could benefit birds in Pennsylvania (Pabian and Brittingham 2007; Pabian et al. 2012), and documented a positive effect of liming on snail abundance in the forest floor and on songbird abundance in northern hardwood forests. Positive relationships between liming and bird reproductive success have been also observed in Europe (Tilgar et al. 2002).

No studies have evaluated the effects of liming on mammals in northeastern North America. However, Pabian et al. (2012) noted that the increase in forb abundance and nutrient content of these plants following liming could improve forage quality and quantity for deer (Cervidae). In the UK, Shore and Mackenzie (1993) mentioned that liming may adversely affect some shrew species in non-forested habitats, but predicted that the effect would be short-lived. In Sweden, Frank (1998) suggested that moose diet was affected by liming, causing diseases similar to a copper deficiency or a molybdenosis-type syndrome.

5. Liming effects on aquatic habitats

Direct application of calcium materials to surface waters has long been used to restore acidic lakes and streams; an extensive literature documents these effects (see reviews by Clair and Hindar 2005 and Mant et al. 2013). As our focus is on forest liming, we examined the subset of aquatic studies in which carbonate materials (e.g., $CaCO_3$, $CaMg(CO_3)_2$) were applied to watershed soils, and particularly those located in the North American hardwood forest biome.

Carbonate materials added to watersheds are largely retained in surficial soil layers. This results in a gradual release of calcium and a long-term source of buffering to receiving waters. For example, Cho et al. (2012) reported that 98% of added calcium (as calcium silicate) was retained over 6 years following addition to a northern hardwood forest in the White Mountains of New Hampshire. Significant increases in calcium and acid neutralizing capacity were observed in both the soil and the stream water following whole-catchment application, although the magnitude of response varied with topography and soil depth. Changes in soil solution chemistry were greater in areas with thin soils and resulted in saturated soil conditions, especially where the water table was high. As calcium concentrations are a principal determinant of the acid-base status of surface waters, greater export from limed soils clearly benefits water quality in regions experiencing soil base cation depletion.

Catchment liming is particularly advantageous where direct applications to surface waters have short-lasting effects due to short water residence time. At a site in Norway, suitable water quality conditions were maintained for over 20 years following catchment liming (Traaen et al. 1997). In the United States, one of the most detailed and long-term liming studies on aquatic biota was conducted at Woods Lake, in the northern hardwood forest of the Adirondack Mountains (New York, Driscoll et al. 1996). Lake chemistry monitoring revealed that treating the watershed resulted in a more gradual change in pH, acid neutralizing capacity, and calcium than direct additions of $CaCO_3$ in the lake, which were characterized by rapid increases in pH and subsequent re-acidification. Thus, catchment application likely precludes the deleterious effects on biota that may occur when base materials are added directly, causing abrupt changes in pH (Driscoll et al. 1987; Bukaveckas 1989). Most importantly, watershed calcium treatments decrease the export of trace metals (e.g., aluminium) to surface waters (Cirmo and Driscoll 1996; Cho et al. 2012). Aluminum toxicity is known as one of the main causes of deleterious effects on aquatic biota (Cronan and Schofield 1979; Reuss and Johnson 1986). Watershed treatment also reduces the occurrence of episodic acidification in streams and lake littoral regions, during runoff events associated with rainfall and snowmelt. Mitigating these effects is particularly important for the reproductive success of certain fish species (e.g., brook trout) since it allows them to establish viable populations. Cho et al. (2009) showed that a positive acid-neutralizing capacity was maintained in a stream draining its calcium-treated catchment, whereas average values were negative in the nearby control stream. They estimated that the calcium addition contributed ~20% of stream calcium during these events. Overall, watershed liming has been shown to successfully reduce aluminum leaching from soils, and to increase the pH and acid buffering capacity of surface waters, thereby preventing potentially toxic conditions.

Changes in water chemistry following liming have been shown to have positive effects on community diversity for a wide range of biota including plankton, aquatic vegetation, and communities of macroinvertebrates, fish, and amphibians (Keller et al. 1990; Molot et al. 1990; Beattie and Tyler-Jones 1992; Bellemakers and van Dam 1992; Beattie et al. 1993; Weiher et al. 1994; Dickson et al. 1995; Henrikson et al. 1995; Larsson, 1995; Roelofs et al. 1995; Svenson et al. 1995; Bukaveckas and Shaw 1997). The result is the transformation of simple communities dominated by acid-tolerant species to more complex food webs which have comparable diversity to circum-neutral lakes. It is important to note, however, that re-colonizing species may not be the same as those present prior to acidification; as a result, post-liming assemblages may differ from communities that developed over the long history prior to acidification (Yan et al. 2003; Clair and Hindar 2005). While the effects of liming on food web structure are well documented, studies on changes in functional properties (e.g., primary production, zooplankton grazing) are less common (Sierszen and Frost 1990; Schindler et al. 1991; Bukaveckas 1993). The lack of response in functional properties may reflect functional redundancy within communities as well as complex dynamics arising from post-liming species interactions (Locke and Sprules 1994).

Fish, especially commercially and genetically important strains, are often a focal point of restoration. Consequently, there is an extensive literature on their responses to liming. In northeastern North America, many studies have focused on Salmonidae and have documented improvements in survivorship growth and recruitment (Schofield and Keleher 1996; Clayton et al. 1998; Hudy et al. 2000; McClurg et al. 2007). However, population-level responses were often unstable, particularly in the first years following liming. In addition, the combined effects of chemical and biological manipulation (e.g., fish stocking) may result in unstable populations due to complex interactions within food webs (Holmgren 2001; Clair and Hindar 2005; Guhrén et al. 2007). In their review of liming effects on rivers, Mant et al. (2013) found that the response of fish populations was highly variable, but that all rivers that had been limed for more than 7.5 years showed a positive response.

6. Conclusion

As the abundance and diversity of many flora and fauna species are closely and positively correlated to soil and water pH and calcium, it is likely that acid rain has negatively influenced some of these populations in sugar maple dominated stands with low soil buffering capacity (Driscoll et al. 2001). In recent decades, liming has been effectively used to mitigate acid deposition in these forests. Although some detrimental effects of liming have been observed on some forest and lake components, current literature suggests that a low to moderate lime application every 25–30 years will have no or only positive effects on key fauna or target flora in these forest ecosystems. Moreover, liming could profit some forest amphibians that reproduce in acidified forest ponds, streams, or lakes. The few long-term liming experiments in northern hardwood forests have shown that liming slowed or did not increase the decomposition rate of organic matter in the forest floor. Furthermore, limed northern hardwood forests could potentially suffer long-term negative effects resulting from earthworm invasions brought on by soil conditions more favorable to certain earthworm species.

Our review of the liming literature for aquatic habitats, in agreement with prior published reviews (Clair and Hindar 2005; Mant et al. 2013), indicates that liming undertaken to improve forest health is likely to improve water quality and yield benefits for nearby aquatic habitats. Provided that water quality improvements are sustained, these will allow the establishment of acid-sensitive species and counteract effects associated with atmospheric deposition. In conjunction with concurrent efforts to reduce atmospheric emissions, liming should speed biological recovery in affected areas.

Although the species composition in limed stands will likely differ from the previous forest, this treatment could mitigate the effect of soil acidification, and consequently, increase the sustainability of sugar maple in these ecosystems. However, more studies are needed to document the long-term effect of liming, particularly on unstudied key species, to determine whether liming is an appropriate treatment, based on management and conservation objectives. A final choice of liming as a mitigation tool should not only weigh the potentially negative effects of this treatment, but should also consider ecological components that could be lost or never recovered if an acidified forest is not limed.

Acknowledgements

This research was supported by the Ministère des Ressources naturelles du Québec (project no. 112310063). We wish to thank Denise Tousignant for reviewing the manuscript.

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