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Effects of environmental pollution on forest soil microflora – a review

Hannu Fritze

The article is a literature review focusing on the reaction of soil respiration, litter decomposition and microflora of forest soils to various pollutants like acidic deposition, heavy metals and unusual high amounts of basic cations.

Keywords: pollution, acidification, heavy metals, lime, forest soil, soil microflora, soil respiration, litter decomposition, fungal hyphae.

Correspondence: Hannu Fritze, Department of Forest Ecology, Finnish Forest Research Institute, P.O. Box 18, SF-01301 Vantaa, Finland.

Introduction

Environmental pollution is a serious problem that affects both aquatic and terrestrial ecosystems and most people in the industrialized world should, by now, be aware of the general term "acid rain". From the recently published book "Acidification in Finland" (Kauppi et al. 1990) the term "acidification" can be interpreted as describing the deposition of atmospheric sulphurous and nitrogenous acids into terrestrial or aquatic environments. However "acidification" is also used in a wider context to deal with various different anthropogenic compounds that enter the ecosystem.

The most active studies of pollution in the terrestrial ecosystem has mainly been concentrated on the separate or combined effects of sulphur, nitrogen and heavy metals on forest vegetation. Considerable research activity has also lately been focused on the effects of ozone in this environment. The forest ecosystem is composed of two active compartments representing productivity and decomposition. The organic matter of forest soil, containing both living and dead components, constitutes a major supplier of nutrients to the ecosystem (Paul and Clark 1989). All organic material deposited on or in the soil will be decomposed and mineralized mainly through the activities of fungi, bacteria and soil animals, of which the fungi are considered to be the most important decomposers as they are metabolically the most active (Anderson and Domsch 1975). Soil microorganisms are key components in the biogeochemical cycling of various chemical elements, and hence of prime importance in maintaining the fertility of terrestrial habitats. Consequently, factors altering the rates of microbial processes in soil may be of importance for forest productivity (see Bååth 1985).

This review deals with the response of soil microbes and their activity in coniferous forests exposed to various pollutants which include acidic deposition, heavy metals or an unusual high concentration of alkaline cations. The review has been restricted to the microbiology of bulk soil and concentrates on the analyses of soil respiration, litter degradation, fungal hyphae, microbial biomass and its constituents. The possible effects of the pollutants on rhizosphere microorganisms, mycorrhiza, and on soil enzymatic activity are not included.

Effect of anthropogenic acids on soil microorganisms

Acidification of the environment is now recognized as a very serious environmental problem. Anthropogenic emission of large quantities of oxides of sulphur and nitrogen into the atmosphere from industry, fuel combustion and traffic are the main causes of acidification (Rodhe 1989).

The naturally acidic condition of the podzolized northern coniferous forest soils is mainly due to the presence of organic acids which are derived from decomposing litter of trees and ground vegetation (Nihlgård et al. 1988). The

introduction of additional acids such as sulphuric and nitric acids from dry and wet deposition will further increase soil acidity, leading to the leaching of alkali metal (Ca, Mg, K and Na) cations from the uppermost soil horizons as a result of displacement by hydrogen ions. The higher hydrogen ion concentration also accounts for the solubilization of aluminum hydroxides (Marion et al. 1976) which can lead to toxic concentrations of free Al in the soil. The soil acidification–aluminum toxicity hypothesis was originally presented by Ulrich et al. (1979) but it is probably not only the toxicity of aluminum and nutrient leaching which influences soil life, the anionic constituent of the acids may also be an important factor (Bewley and Stotzky 1983a).

Soil respiration

Respiration, which is considered to represent overall soil microbial activity, is the most commonly analyzed variable in environmental pollution studies. Artificial acidification has resulted in decreased rates of basal (no energy substance added to the soil) or induced (energy substance added to the soil) respiration in many laboratory experiments (Tamm 1976, Bryant et al. 1979, Strayer and Alexander 1981, Francis 1982, Bewley and Stotzky 1983b, Killham et al. 1983, Klein et al. 1984, Bitton and Boylan 1985, Persson et al. 1989, Salonius 1990). Low pH (approx. pH 2) treatments were in some cases required to depress soil respiration, but alternatively weaker treatments have been shown to stimulate respiration (Killham et al. 1983). Different soil types also reacted differently. An identical acidic treatment was found to alternatively decrease, increase or have no influence on CO_2 evolution when applied to different soils (Bitton and Boylan 1985).

Artificial acidification in the field has also yielded conflicting results. Decreased rates of respiration were reported by Bååth et al. (1979), Bååth et al. (1984), and Lohm et al. (1984) but Johnson and Todd (1984), Bitton et al. (1985), Will et al. (1986) and Nohrstedt (1987) observed no changes, even though in the latter study the artificially acidified rainwater had been supplemented with heavy metals.

In both natural coniferous and deciduous forests exposed to anthropogenic acidic pollution from various sources a decrease in soil respiration has been reported (Bryant et al. 1979, Prescott and Parkinson 1985, Bewley and Parkinson 1985, Bewley and Parkinson 1986a and b, Maynard et al. 1986, Schäfer 1987, Fritze 1987). In contrast to these studies Nohrstedt (1985) found no respiratory changes in soil samples from an area polluted with sulphur dioxide taken 16 years after closure of the emission source. Visser and Parkinson (1989) also could not demonstrate any differences in soil respiration from samples taken along a gradient (750 m) from a point source polluting the environment with elemental sulphur; the contribution of elemental sulphur to soil acidity was likely to have resulted from the elevated microbial oxidation of S_0 to SO_4 by *Thiobacillus* species.

In the case of sulphuric acid, toxicity can be due to the acid itself, to the resulted leaching of essential nutrients and from the increased aluminium solubility as stated earlier. The same can also be said for nitric acid although nitrogen is usually a growth-limiting nutrient in the forest ecosystem (Kenk and Fischer 1988) and aerial deposition can also be in the form of ammonia. Excess nitrogen in the forest ecosystem is known to induce the following detrimental effects: deficiency of other macro- and micronutrients due to increased plant growth and hence a greater demand for these other elements, increased susceptibility to frost, inhibition or necrosis of mycorrhizae, increased susceptibility to fungal root pathogens, changes in root-shoot ratios, and altered patterns of nitrification, denitrification, and also possibly nitrogen fixation (Hinrichsen 1986). Additionally, ammonia deposition contributes to soil acidification. Nitrifying microorganisms release 2 mol of hydrogen-ions per 1 mol ammonium and during ammonium uptake by plants one mol of hydrogen-ions is released per mol of ammonia (Grennfelt and Hultberg 1986). If the forest ecosystem does not receive sufficient nitrogen from the air and no ammonium or nitrate is leached from the soil then the sum of the produced and consumed hydrogen-ions will be zero during any nitrogen transformation (Grennfelt and Hultberg 1986, Martikainen 1988). This still holds if the amount of aerially deposited ammonium is equal to the amount of similarly deposited nitrate if no nitrogen is leached from the soil.

Nitrate is readily leached from the soil (Grennfelt and Hultberg 1986) and contributes to acidification in two ways: firstly, associated neutralizing basic cations are concomitantly leached with nitrate and secondly, denitrification by microorganisms and plant uptake of nitrate are hydrogen-ion consuming reactions. Hence the excess hydrogen-ions produced from nitrification and plant ammonia uptake are retained during leached nitrate loss from the soil. Fertilization with NH_4NO_3-N or urea-N have resulted in lower soil respiration (Söderström et al. 1983, Nohrstedt et al. 1989). In the former study the pH reduction in the soil, due to NH_4NO_3 addition, was only partly attributed to the lowered respiratory activity the and remaining decrease was due to the addition of nitrogen itself.

Litter decomposition

In order to be able to compare the investigations cited below an estimate of annual load of protons on a soil area basis was in each case calculated from the experimental designs given by the authors. The values were calculated on the basis of yearly application of the acidic loads. This may overestimate the proton loads of some treatments and underestimate others but it makes comparison of treatments possible. For each estimate it was assumed that acids were totally dissociated in water and that one mol of ammonium contributed to two moles of protons via nitrification. The neutralizing nature of ammonia was not considered.

Mass loss studies

In forest ecosystems the major part of the net primary production results in the formation of litter (Paul and Clark 1989) which through microbial activity in soil is decomposed and subsequently released as nutrients for plant growth (see Bååth 1985).

The effects of artificial acidification on litter decomposition has been widely reported (Bååth et al. 1980, Hovland et al. 1980, Hågvar and Kjøndal 1981, Lee and Weber 1983, Berg 1986a and b). Lee and Weber (1983) reported an enhanced decomposition of hardwood leaf litter when treated with dilute sulphuric acid of pH 3.5, but not with acid of pH 3.0 equaling 12×10^3 mol H^+ ha^{-1} $year^{-1}$. In their experiment the acid was applied three days a week for a period of 408 days. Similar results where retarded decomposition rates were seen only for the highest acid application level was reported for spruce needles (Hovland et al. 1980) and birch leaves (Hågvar and Kjøndal 1981). Berg (1986a and b) detected a decrease in the decomposition rate of needle litter in Scots pine (*Pinus sylvestris*) and Norway spruce (*Picea abies*) stands as a result of artificial acidification. In the former study, Berg had applied dilute sulphuric acid, equivalent to 32 and 96 kg SO_4-S ha^{-1} over a 6 months period which corresponded to an annual input of 4×10^3 and 12×10^3 mol H^+ ha^{-1}, respectively.

Recently Neuvonen et al. (1990), reported reduced decomposition of Scots pine needle and mountain birch (*Betula pubescens ssp. tortuosa*) leaf litter in field experiments acidified with a mixture of H_2SO_4 and HNO_3 at pH 3 and 4. The pH 3 treatment in their experiment received a 15-fold sulphur load and a 11-fold nitrogen load compared to the ambient air pollution of the area: 1.52 kg SO_4-S ha^{-1} $year^{-1}$, 0.32 kg NH_4-N ha^{-1} $year^{-1}$, and 0.40 kg NO_3-N ha^{-1} $year^{-1}$, corresponding to 0.095×10^3, 0.046×10^3 and 0.029×10^3 mol H^+ ha^{-1} $year^{-1}$, respectively. The litter bags containing the needles or leaves had been mounted onto the litter layer of the forest floor.

In an area surrounding a point source with large emissions of SO_2 (36 t per day) and elemental sulphur, retarded decomposition of pine needles towards the point source was also observed (Prescott and Parkinson 1985). In contrast to this Fritze (1988) reported that urban pollution (emission data: 26000 t SO_2 and 15000 t NO_x per year) had no effect on the decomposition rate of pine or spruce needles incubated in the soil humus for one year.

Nutrient dynamics in decomposing litter

The release of the nutrients N, P, S, K, Ca, Mg, and Mn from the decomposing needle litter has been studied by Staaf and Berg (1982). They showed a linear correlation between accumulated weight loss and nutrient concentration which suggest that N, P, and S were immobilized during the decomposition of needle litter. Of the other nutrients, K and Mg were first released but were then slightly immobilized in the later decomposition phase (> 50 % mass loss), whereas Ca and Mn showed a constant release pattern for the whole decomposition process.

When soil mounted with coniferous litter bags was acidified with 32 or 96 kg SO_4-S ha^{-1}, corresponding to 2×10^3 mol and 6×10^3 mol H^+ ha^{-1} (equal to an annual input of 4×10^3 and 12×10^3 mol of protons per hectare, respectively), a greater release of Mn was observed in the decomposing litter but a greater release of Mg

was only measured in the higher S treatment (Berg 1986a). Hovland et al. (1980), using laboratory lysimeters, demonstrated the leaching of K, Mg, Mn and Ca from Norway spruce needles treated with different amounts of distilled water acidified to pH 3 or 2 with sulphuric acid for 16 or 38 weeks. The treatments equaled annual depositions of 12×10^3, 24×10^3, 120×10^3, and 240×10^3 mol H^+ ha^{-1} $year^{-1}$, respectively. The leaching of all cations became apparent in the 24×10^3 mol treatment (Hovland et al. 1980).

Brown (1985) demonstrated the leaching of Ca, Mg, K and Mn from bracken litter when acidified with dilute sulphuric acid, pH 3, (equivalent to 240 kg S ha^{-1} $year^{-1}$, corresponding to 15×10^3 mol H^+) over a period of five years. Similar results were obtained by Skiba and Cresser (1986) in an experiment where simulated rain, acidified with sulphuric acid to pH 3 (7.1×10^3 mol H^+ ha^{-1} $year^{-1}$), was applied to Sitka spruce litter during a 3 month incubation on a soil from a spruce stand. The litter had been preincubated with sulphuric acid pH 3 (0.7×10^3 mol H^+ ha^{-1} $year^{-1}$) for 7 months. In the study of Lee and Weber (1983) Mn was the only element that was leached from ten different hardwood leaf litters when treated with dilute sulphuric acid, pH 3, for 408 days (12×10^3 mol H^+ ha^{-1} $year^{-1}$).

In an urban area, where the anthropogenic emissions of oxidized sulphur and nitrogen compounds had no effect on the decomposition rate of coniferous litter burried into the humus layer, a higher rate of Mn release from the litter was obvious in the affected area as when compared to the reference area (Fritze 1988).

Microflora

It is generally believed that the fungal component of the microbial biomass is greater than the bacterial component in acid forest soils (Paul and Clark 1989). The application of artificial acid rain decreases both the bacterial and the fungal biomass of the soil. Staining with fluorescein diacetate (FDA) can be used to express microbial activity in soil; Bååth et al. (1979) reported both a decrease in the length of fungal hyphae and bacterial counts in response to the artificial acidification of a plantation of young pine seedlings which had received sulphuric acid acidified water at pH 3 and pH 2 over a period of four years. The pH 3 treatment resulted in the addition of 40 kg of SO_4-S ha^{-1} during 5 months of the year, equivalent to 6×10^3 mol H^+ ha^{-1} $year^{-1}$. A significant decrease of FDA-active bacteria was only detected in the pH 2 treatment. No effect, however, could be seen on the total length of fungal hyphae.

In a further experiment Bååth et al. (1980) showed that repetitive application of 150 kg (3×10^3 mol H^+) H_2SO_4 ha^{-1} for six subsequent years reduced the FDA-active fungal hyphal length in the humus layer when given as a single annual dose. At greater depth, however, the decrease was also apparent with an application of 50 kg (1×10^3 mol H^+). The total length of fungal mycelium was not influenced by any of the acid treatments in the humus layer and was even found to increase in the lower soil profile. The number (microscopical counts) of acridine orange stained bacteria in the humus layer was reduced in the 150 kg treatment. However, four years later, the amount of FDA-active fungal hyphae was significantly smaller in the humus layer, as compared to the controls, in all the treatments even though no acid was applied between the two studies (Bååth et al. 1984). Similarly a changed composition of microfungal species in response to the 150 kg treatment was found (Bååth et al. 1984).

Persson et al. (1989) reported on two laboratory experiments, where in one the FDA-active fungal hyphae decreased in response to the acid treatment and in the other experiment no changes occured. The acid treatments in the two experiments were similar and corresponded to 38 kg (0.780×10^3 mol H^+) and 150 kg (3×10^3 mol H^+) H_2SO_4 ha^{-1} during a 55 day period. This would correspond to an annual application of 5.1×10^3 and 20×10^3 mol H^+ ha^{-1} $year^{-1}$, respectively. The reason for this observation can be explained by the different C/N-ratios of the soils. According to the results of Persson et al. (1989), a high C/N-ratio seemed to make the FDA-active fungal hyphae more independent of changes in pH than a low C/N-ratio soil.

Application of artificial acid rain (identical rates of a H_2SO_4/HNO_3 pH 3 mixture as given above for Neuvonen et al. 1990) had no effect on FDA-active hyphae, total length of fungal hyphae or on AO-stained bacteria but a significant decrease in the numbers of bacteria utilizing different organic compounds was observed (Kytöviita et al. 1990). Bååth et al. (1980) also observed that the acid treatments caused marked changes in the physiological characteristics of the soil bacterial population. In a laboratory experiment simulating a 3.2 m high polluted snowpack, acidified to pH 2.3 (equivalent to 880×10^3 H^+ ha^{-1} $year^{-1}$), melting into the soil, Thompson et al. (1987a) demonstrated a negative effect on the number of bacteria in the humus fraction and subsequently alterations in their physiological characteristics (Thompson et al. 1987b).

In the forest environment near a sour gas processing plant (emitting 36 t SO_2 day^{-1} in addition to elemental sulphur deposition), a significant reduction in total heterotrophic counts of bacteria and starch-utilizing bacteria in the humus layer could be demonstrated at a distance of 2.8 km from the plant as compared to a control site (Bewley and Parkinson 1984). Later a reduction of microbial biomass (Bewley and Parkinson 1985) and changes in the microflora able to decompose vanillin (Bewley and Parkinson 1986a) were reported. Maynard et al. (1986) reported a non-significant decrease of bacteria in a forest soil acidified (pH 2.6) by the action of *Thiobacilli* oxidizing stored elemental sulphur and at sites close to the former, Visser and Parkinson (1989) reported reduced soil microbial biomass. In the surroundings of an oil refinery, mainly emitting sulphur dioxide (24000 t yr^{-1}), only the fungal hyphal length was found to decrease due to the pollution, whereas no significant differences in soil respiration, microbial biomass or the decomposition rate of needle litter could be detected (Fritze et al. 1992).

Bewley and Stotzky (1983b) showed in a laboratory experiment that the toxic effect of artificial acid rain on the fungal growth rate and soil respiration could be overcome by additions of montmorillonite to the soil. This single experiment highlights the difficulties in comparing results obtained in different ecosystems on various soil types receiving different amounts of acids. To enable comparison of the above studies and for the future a much better description of the physicochemical characteristics of the recipient soil is needed when assessing the impact of acid precipitation on terrestrial microbial communities. Soil chemical variables related to acidification such as exchangeable acidity, base saturation, and cation exchange capacity should be provided.

Many of the studies with experimental acid rain involved acid loads exceeding the present deposition in Finland. The southern part of Finland receives the highest sulphate, nitrate and ammonium depositions and general levels decrease towards the north. According to the latest calculations for southern Finland the annual SO_4-S deposition in 1987 was 12 kg per hectare, equal to 0.75×10^3 mol H^+ (Järvinen and Vänni 1990, Tuovinen et al. 1990). In 1985 the amounts were 2.5 kg each for oxidized (0.18×10^3 mol H^+) and reduced nitrogen (0.36×10^3 mol H^+) per hectare (Tuovinen et al. 1990). This suggests a potential annual input of about 1.3×10^3 mol protons per hectare in a year. Against this background, many of the experiments cited above have only an indicative value.

In conclusion, acid deposition may be expected to reduce the activity of the soil microbial flora, to affect the species composition and to reduce the soil microbial biomass.

Effect of heavy metals on soil microorganisms

Heavy metals in the atmosphere originate from the combustion of fossil fuels, from the smelting of metal ores, and from the use of leaded petrol. From studies made in the Nordic Countries it is known that heavy metals are transported over long distances and that, in general, the deposition in the south is higher than in the north (Rühling et al. 1987). Many heavy metals are essential micronutrients for microorganisms, plants and animals, including humans but at elevated concentrations these metals are toxic to life. The mechanisms of metal toxicity can be divided into three categories, those (1) blocking essential functional groups of biological molecules, (2) displacing an essential metal ion in biomolecules, or (3) modifying the active configuration of biomolecules (Collins and Stotzky 1989). The detrimental effect of heavy metal pollution on soil microbial processes has been reviewed by Duxbury (1985) and Bååth (1989).

Soil respiration

In laboratory experiments the addition of heavy metals such as Pb (Doelman and Haanstra 1979), or Cr, Cd, Cu, Zn, and Mn (Chang and Broadbent 1981) to soil samples caused depressed respiration. Doelman and Haanstra (1979) observed in a sandy soil a 15 % decrease in soil respiration when amended with 375 µg of Pb g^{-1} dry soil, the lowest concentration used, whereas a clay soil required 1500 µg of Pb g^{-1} dry soil for the same inhibition. A peat soil showed no effects even at the highest concentration of Pb (7500 µg of Pb g^{-1} dry soil). Differential respiration responses

with respect to soil type was reported following the addition of Cd, Cr, Cu, Pb, Ni, or Zn (Doelman and Haanstra 1984) and Cd to the soils (Reber 1989). In the latter study, the addition of Cd at a rate of over 50 µg g^{-1} dry soil had the greatest inhibitory effect on substrate-induced respiration in a phaeosem (loamy silt) followed by a neutral sandy soil. This implies that the toxic effect of heavy metals is also controlled by abiotic soil factors other than the mineral constituents (Doelman and Haanstra 1984, Collins and Stotzky 1989).

Addition of Zn to a sandy clay loam (1000 µg of Zn g^{-1} dry soil) showed a greater influence on bacterial as compared to the fungal respiration and a concomitant reduction in the number of the viable bacteria was observed (Ohya et al. 1985). Basal respiration rate was shown to be the variable most inhibited by heavy metals when compared with substrate-induced respiration, lag time before the exponential increase of the soil respiration rate and the specific respiration increment during the exponential phase (Nordgren et al. 1988).

Field studies at forest sites surrounding smelters agree with the laboratory studies. Respiration of soils decreased with increasing metal content containing up to 2600 µg of Cu and 1900 µg Ni g^{-1} dry soil (Freedman and Hutchinson 1980) or 3600 µg of Cu and 360 µg Ni cm^{-3} dry soil (Fritze et al. 1989) near Cu-Ni smelters and up to 20000 µg of Cu, 20000 µg of Zn, and 1000 µg of Pb g^{-1} dry soil near a brass mill (Nordgren et al. 1983). Lower soil respiration rates were also reported in the surroundings of a smelter emitting many different heavy metals (Fe, Pb, Zn, As, Cu, V, Ni, Cr) where the soil near the smelter contained approximately 6000 µg of Fe and Pb, 2600 µg of Zn, 2000 µg of Cu, 1000 µg of As, and between 10 and 50 µg of Cr, V and Ni g^{-1} dry soil (Nordgren et al. 1986). In the former study the respiration increased with increasing distance from the smelter and concomitantly with decreasing soil metal content.

There are only a few studies where heavy metals had no effect on soil respiration. Nohrstedt (1987) contaminated forest sites with acidified water containing elevated cadmium and copper concentrations representing up to a hundred times the normal deposition. The normal Cd and Cu deposition of that area was 50 µg and 500 µg m^{-2} 5 $months^{-1}$, respectively. No effect on the soil respiration rate was seen. The explanation could be that the heavy metals did not reach the toxic level for this soil, or the metals were washed to deeper soil horizons due to the acidified water. The evolution of CO_2 from decaying needles treated with acidified water, pH 3, enriched with different heavy metals showed additional repression for Pb and Zn at concentrations of 5000 µg and 10000 µg l^{-1} but not for Al or Cu at the same application levels (Moloney et al. 1983).

Litter decomposition

Litter decomposition is affected by heavy metal pollution. Accumulation of litter on the forest floor in response to metallic pollution, which affects the decomposition rate, has been reported (Williams et al. 1977, Strojan 1978, Coughtrey et al. 1979, Freedman and Hutchinson 1980, Fritze et al. 1989, Berg et al. 1991). Application of Cu and Ni to a leaf litter homogenate in the laboratory caused a depressed rate of dry weight loss after the application had exceeded 500 µg of both Cu, and Ni g^{-1} dry litter (Freedman and Hutchinson 1980). Berg et al. (1991) indicated that the lignin decomposition of the litter was more sensitive to metallic pollution than decomposition of whole litter and was affected further away from the pollution source. Ohtonen et al. (1990) showed that the heavy metals emitted from a Cu-Ni smelter were enriched within the decaying needles and that the increased rate of manganese release paralleled the decreased rates of litter decomposition towards the smelter.

Microflora

The effect of heavy metals on soil microbial communities is also apparent. Bisessar (1982) found 28000 µg of Pb, 1000 µg of As, 600 µg of Cu, and 150 µg of Cd g^{-1} dry soil near a secondary Pb smelter. The amounts of these metals decreased with increasing distance from the smelter and showed a negative correlation with the abundance of bacteria including actinomycetes, fungi, nematodes, and earthworms (Bisessar 1982). Ohya et al. (1986) reported a changed bacterial flora, as characterized by the fatty acid composition of the isolates, when soil was amended with 1000 µg of Zn g^{-1} dry soil in the laboratory. However, field studies at urbanly polluted sites, contaminated mainly with Zn and Pb ranging between 50–4700 µg and 3–1500 µg g^{-1} dry soil, respectively, indicated that the microbial biomass was not affected by increasing Pb and Zn pollution although respiration was depressed (Ohya et al. 1988). Brookes and McGrath (1984) reported reduced microbial biomass in soils polluted with Cu, Ni, Cd, Pb, Cr, and Zn. The extractable Cu and Ni concentrations of the soil, respectively, reached 100 µg and 10 µg g^{-1} dry soil, but the concentration of the other heavy metals affecting the microbial biomass was not reported (Brookes and McGrath 1984). Soil artificially polluted with Cu (100–1600 µg of Cu g^{-1} dry soil) showed changes in fungal flora with increasing copper concentrations proving that for some species the pollution was beneficial whereas for others it was toxic (Yamamoto et al. 1985).

Alterations to the fungal flora resulting from metal deposition has also been observed in field studies. The soil microfungal community structure responded to metal deposition near a smelter contaminating the environment with mainly Cu and Zn (Nordgren et al. 1985). A survey of the 26 most common soil microfungal taxa indicated a changed community structure at soil Cu concentrations around 1000 µg g^{-1} dry soil. According to the study the most common coniferous soil microfungi disappeared close to the mill, while fungi that became dominant in the polluted sites are normally rarely isolated in background coniferous forest soils. Species isolated from the polluted sites were usually tolerant to Cu (Arnebrant et al. 1987). Additionally the length of fungal hyphae, both active and total, was reduced by the pollution (Nordgren et al. 1983). This was shown also by Fritze et al. (1989) for the total fungal hyphal length near a Cu-Ni smelter.

In another study near a smelter with a more complex emission pattern (mainly As, Cd, Cu, Pb, and Zn) both the total and the FDA-active stained mycelial lengths in the soil mor horizon remained unchanged with increasing pollution level (Nordgren et al. 1986). However, changes to the fungal community were reflected in the amount and species diversity of Basidiomycetes on the basis of fruitbody diversity and numbers (Rühling and Söderström 1990). The pollution caused a reduction in both the number of species and fruitbodies. At these study sites the pollution also reduced the numbers of bacterial groups able to utilize different organic compounds (Nordgren et al. 1986).

Summing up, heavy metal pollution is a special problem of greater local importance than S and N depositions. All the studies strongly indicate that heavy metals negatively affect the soil microbial community and its activity. The critical load for each metal is different and is strongly dependent on the soil texture and its physicochemical properties (Collins and Stotzky 1989). Collins and Stotzky (1989) attributed the following physicochemical soil variables in having an influence on the toxicity of a heavy metal: (1) pH, (2) oxidation-reduction potential, (3) inorganic anions, (4) inorganic cations, (5) water hardness, (6) clay minerals, (7) organic matter, and (8) temperature.

Effect of base cations on soil microorganisms

Alkaline deposition is often not regarded as a pollution problem but it is in two respects strongly connected with soil acidification. Alkaline substances neutralize some fractions of the acids, and atmospheric deposition of Mg, Ca, Na, and K may be a considerable source of new base cations to a soil ecosystem depleted of them following acidic precipitation (Anttila 1990). Alkaline aerosols are introduced into the environment from open sources such as traffic-raised dust, wind erosion, agriculture and sea salts or from industrial sources e.g. fuel combustion, kraft- and sulphite-pulping, clay products manufacture, cement and concrete processing, solid waste disposal and iron and steel manufacture (Anttila 1990). In Finland deposition values of Ca and Mg show a clear decreasing gradients towards the north with a maximum in southeastern Finland (Järvinen and Vänni 1990). Fritze (1991) published an article on the response of the soil microflora receiving continuous alkaline deposition. As no other soil microbial research has, to my knowledge, been done in areas receiving unusually high depositions of alkaline dust, the effects of liming on forest soil microbiology are summarized, as the effects of liming might be considered analogous to those of alkaline deposition.

The practice of liming acidic forest soils was seen as a measure for counteracting soil acidification by raising the base saturation of the humus soil and concomitantly the soil pH. The following lime applications performed in the studies cited below have been calculated on a hectare basis: 1960 kg (Lohm et al. 1984), 4000 kg (Zelles et al. 1987), 3000 kg (Persson et al. 1989), 7500 kg (Shah et al. 1990), 1960 kg (Bååth et al. 1980), 640 kg (Berg 1986b), 2000,

4000, and 8000 kg (Hojito et al. 1987), and 1000 kg (Carter 1986). Haynes and Swift (1988) amended 1.8 mg and 4.3 mg of $Ca(OH)_2$ g^{-1} dry soil, respectively, in their laboratory experiment.

Soil respiration

Liming of acid forest soils is known to increase soil respiration. Laboratory studies have shown that liming of mor humus initially induced an intense evolution of CO_2 which then declined with time (Persson 1988). In certain field studies (see Söderström 1984, Lohm et al. 1984) the respiration rate in limed mor humus remained higher than in untreated plots for as long as 2–7 years after liming. Elevated respiration in limed forest soils has also been reported by Zelles et al. (1987), Haynes and Swift (1988), Persson et al. (1989), and Shah et al. (1990). Also in the study with the soil receiving continuous alkaline deposition elevated soil respiration was detected (Fritze 1991). A greater supply of soluble C sources following the elevation of soil pH has been proposed to explain the increases in soil respiration after liming (Persson et al. 1989).

Litter decomposition

The effect of liming on litter decomposition has not been greatly studied. Bååth et al. (1980) and Berg (1986b) could not show any significant stimulatory or inhibitory effects of lime application on mass loss of coniferous needle litter. Neither could Fritze (1991) demonstrate any effect of alkaline deposition on the decomposition rate of pine needle litter. Müller and Berg (1988) showed that organic residues, derived from clover roots during decomposition, provide a well buffered microenvironment for the decomposing microflora.

Microflora

The bacterial component of the soil microbial communities seem to be most affected by liming. In all plate counting studies increased numbers of bacteria have been recorded following liming (see Söderström 1984, Hojito et al. 1987) although the numbers have remained constant when counted microscopically (Bååth et al. 1980, see Söderström 1984). Persson (1988) suggested that a shift to more culturable bacterial populations rather than an actual increase in the numbers of bacteria was the reason for the observed behaviour.

No significant effects of liming on the FDA-active or total fungal hyphal lengths in the humus horizon of a coniferous forest soil have been found, although increases in the total fungal hyphal length was seen in the B-horizon (Bååth et al. 1980) and in the soil of an acidified grassland (Hojito et al. 1987). Persson et al. (1989) reported on two liming experiments, one of which resulted in decreases of FDA-active hyphae and unchanged total mycelial length and the other in increased FDA-active fungal hyphae but a decrease of total hyphal length. Shah et al. (1990) reported unaltered of fungal hyphal length after liming.

Carter (1986) reported on a higher microbial biomass C in laboratory experiments with limed soil. In a similar study by Haynes and Swift (1988), higher microbial biomass N was also reported in response to liming but four weeks later the biomass N had declined back to control levels. As no bacterial or fungal counts were made in these two studies it cannot be ascertained whether the biomass increase was due to the bacterial or fungal growth. In contrast to these studies Fritze (1991) reported the microbial biomass C and N to decrease or increase a little when expressed per soil volume or organic matter, respectively, due to the alkaline deposition. As there has an increase of the fungal hyphal length been measured in the same area it was concluded that the alkaline deposition did not decrease the fungal biomass but could have affected negatively the bacterial part of the soil microbial biomass (Fritze 1991). In two yet unpublished works it became evident that neither the fungal or the bacterial biomass was affected by the alkaline deposition (Fritze and Bååth 199X, Bååth et al. 199X) when expressed per organic matter content of the soil. The fungal biomass was assessed by the soil extracted amounts of ergosterol and the bacterial biomass by the sum of soil extracted bacterial phospholipid fatty acids (PLFA), respectively. But a change of the microfungal species composition, resembling that of limed soils, characterized the soil of the alkaline deposition affected area (Fritze and Bååth 199X).

Conclusions

There is a great deal of evidence indicating that environmental pollution affects soil microbial activity and community structure. Much of the data originates from experimental designs where high levels of pollutants were applied to the soil under field or laboratory conditions. Furthermore, many were short-term experiments designed to look for large effects. These experiments have an indicative value, but it has to be kept in mind that environmental pollution is a combination of many pollutants, mostly at low concentrations, acting over long periods of time. There is therefore consequently a demand for research performed in natural forest environments polluted with anthropogenic compounds.

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