# **Environmental effects of aluminium**

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## **Abstract**

Aluminium (AI), when present in high concentrations, has for long been recognised as a toxic agent to aquatic freshwater organisms, *Le.* downstream industrial point sources of AI-rich process water. Today **the**  environmental effects of aluminium are mainly a result of acidic precipitation; acidification of catchments **leads**  to increased AI- concentrations in soil solution and freshwaters. Large parts of both the aquatic and terrestrial ecosystems are affected.

In the aquatic environment, aluminium acts as a toxic agent on gill-breathing animals such as fish and invertebrates, by causing loss of plasma- and haemolymph ions leading to osmoregulatory failure. In fish, the inorganic (labile) monomeric species of aluminium reduce the activities of gill enzymes important in the active uptake of ions. Aluminium seems also to accumulate in freshwater invertebrates. Dietary organically complexed aluminium, maybe in synergistic effects with other contaminants, may easily be absorbed and interfere with important metabolic processes in mammals and birds.

The mycorrhiza and fine root systems of terrestrial plants are adversely affected by high levels of inorganic monomeric aluminium. As in the animals, aluminium seems to have its primary effect on enzyme **systems**  important for the uptake of nutrients. Aluminium can accumulate in plants. Aluminium contaminated invertebrates and plants might thus be a link for aluminium to enter into terrestrial food chains.

#### **Introduction**

Acidic waters have been recognised as a problem for freshwater fisheries in certain regions of Norway since the 1920s (Dahl, 1927). Forty years later a link between acidic waters and pH of precipitation was hypothesised (Dannevig, 1959). Aluminium:(Al) as a toxic element in acidic waters was recognised nearly 20 years later (Schofield, 1977, Dickson, 1979). Today the two elements,  $H^+$  (pH) and Al, are still considered to be most important causes of toxicity in freshwater biota (Wood and McDonald, 1987).

The biological significance of Al-speciation and the toxicity of the monomeric labile Al-species on fish was demonstrated by Driscoll *et al.* (1982). Heriksen *et al.*  (1984) demonstrated their relevance during episodic changes in water quality occurring in streams and rivers.

Freshwater invertebrates disappear in acidic waters as a response to low pH and aluminium (Herrmann, 1987a). In addition, bioaccumulation of aluminium in invertebrate prey organisms has been suggested as a possible explanation to the impaired hatching success observed among birds (Nyholm, 1981), as dietary fed A1 accumulates in both birds and mammals.

In the late 1970s, another part of the terrestrial ecosystem, forest trees, was suspected to be threatened by acid rain (Ulrich, 1980). Today, in many of the forested areas of central Europe, trees are dying. Aluminium concentration in the soil solution increases due to soil

acidification. Aluminium has been shown to have a negative effect on the root systems of herbaceous plants and trees, and might thus be one of the factors involved in the forest decline.

This paper reviews the role of aluminium in terrestrial and aquatic ecosystems affected by acid rain.

# Effects of **Aluminium on Vascular Plants at Low pH**

Aluminium is the third most abundant element in the soil, constituting on the average 8% of the minerals. On weathered mineral surfaces A1 exists as oxides **and**  polymeric hydroxides. Under acidic conditions these compounds dissolve to form the hydrated ion Al  $(H_2O)6$ <sup>3+</sup> (written  $Al^{3+}$  for simplicity) or hydrolysis products of this ion. The AI ions bind to cation- exchange sites on soil particles and are thus accessible for plant roots.

 $Al^{3+}$  is the dominant A1 ion at pH < 4.5. Since the beginning of this century,  $Al^{3+}$  has been regarded as an important growth limiting factor in acid soils, together with low pH and lack of macronutrients in such soils. Experiments to elucidate the effects of A1 have been done mainly with herbaceous species (for review, see *e.g.* Foy, 1974). During the last 40 years several studies have **also**  been done with tree species, especially after Ulrich *et al.*  (1980) claimed that A1 ions might be an important **factor**  in the forest decline in central Europe.

Aluminium taken up by roots is mainly found in the mucilage layer on the root tip surface (Horst *et al.,* 1982) and m the walls of the epidermis and cortex cells (Huett and Menary, 1980). In the cell wall pectins, A1 ions compete with  $Ca^{2+}$  for the same absorption sites (Wagatsuma, 1983). Some Al is taken up in the cytoplasm and bound to nucleic acids and acid soluble phosphates (Wagatsuma, 1983). Aluminium is translocated only to a small extent to shoots.

All studies of growth and mineral nutrition in the presence of  $Al^{3+}$  (e.g. Göransson and Eldhuset, 1987) confirm that after AI addition, symptoms are first seen in roots by a reduction in length growth, root thickening and root tip dieback. New side roots are short, thick and brittle. and sometimes loss of geotropism occurs. With prolonged A1 treatment, symptoms are also seen in shoots as growth reduction, yellowing and purpling, wilting, and sometimes loss of apical dominance. Ca and Mg concentrations are also greatly reduced in roots and shoots of Al-treated plants.

 $Al^{3+}$  is also toxic to many fungi. Thus, Al may negatively affect the symbiosis between fungi and roots, the so-called mycorrhiza, where the ecto-mycorrhiza seems most affected (Rolf A. Olsen, personal communication). Mycorrhiza is important for the water and nutrient uptake in plants.

 $Al<sup>3+</sup>$  affects several physiological processes. One irnportant effect is reduced membrane potential in root cells, probably due to A1 interference with the Ca- binding protein, calmodulin (Siegel and Haug, 1983a). A1 reduces the activity of several enzymes, including ATPases important in cation uptake (Siegel and Haug, 1983a, Suhayda and Hang, 1986). The ATPases are activated by Ca- calmodulin and are involved in cation uptake. A1 ions and nutrient cations compete for the same uptake sites on cell surface (Kinraide and Parker, 1987). Increased permeability of non-electrolytes and decreased water permeability in cortex cell membranes, probably due to changes in membrane lipids (Zhao *et al.* 1987), are also a significant response to A1. Aluminium phosphate complexes in mucilage and cell walls reduce P availability (McCormick and Borden, 1974). Inhibition of DNA synthesis in cell nuclei is observed, probably due to Al binding to phosphate groups in DNA (Matsumoto and Morimura, 1980). This may be one reason for inhibited cell division and root growth (Horst *et al.,* 1983).

Plants growing on acid soil may have developed mechanisms for tolerating high  $Al^{3+}$  concentrations in the soil. Such mechanisms might include excretion of organic acids which act as complexing agents for  $Al^{3+}$  (Lee and Foy, 1986), increase of pH outside roots so that A1 is polymerised or precipitated (Taylor and Foy, 1985), and active exclusion of Al outside the plasmalemma of root cells (Wagatsuma and Yamasaku, 1985).

Not only the absolute concentration of  $Al^{3+}$ , but also the molar ratio of  $Al^{3+}$  to other ions in soil or nutrient solution is important for the extent of Al toxicity. This is especially true for  $Ca<sup>2+</sup>$ . Increasing Ca concentration decreases AI toxicity (Rost-Siebert, 1983; Abrahamsen, 1984; Alva *et al.,* 1986a). The activity of other monomeric forms of Al (Alva *et al.,* 1986b) or total monomeric AI (Wright and Wright, 1987) may in some cases be a better measure of Al-toxicity than is the  $Al^{3+}$  concentration.

Experiments to determine toxicity of AI are most often done on young plants growing in nutrient solutions in a growth chamber or glasshouse. Such experiments generally indicate that growth reductions start at external A1 concentrations below 5.4 mg/L in herbaceous species and above 20 mg/L in tree species adapted to growth on acid soils. There are great variations among species. Also, the tolerance threshold for a species is to some extent dependent on which growth parameter is measured. The growth systems mentioned are simple and verifiable, but the results cannot be directly transferred to the field situation.

Most tree roots are situated in the upper soil horizons, *i.e.* the O, A, and the upper B horizon. The total Al concentration increases with soil depth, and in the O and A horizon the organic A1 complexes dominate (Nilsson and Bergkvist, 1983). These are regarded as non-toxic. In the B horizon from acid podzolic forest soils in Scandinavia, total AI concentrations of maximum 5.4 mg/L (typically less than 2.7 mg/L) have ben measured (Abrahamsen, 1983; Nilsson and Landmark, 1986).

On the other hand, in some forest soils in southern Sweden there has been a long term decrease in pH (Hallbäcken and Tamm, 1986), a decrease in Ca/Al ratio (Tyler, 1987), and an increase in inorganic AI concentration (Bergkvist, 1987). The same tendency has been seen in West Germany (Ulrich, 1980, 1981).

Among the common forest tree species in Scandinavia, Scots pine *(Pinus sylvestris)* seems to be more tolerant to A1 than white birch *(Betula pendula),* which is again more tolerant than Norway spruce *(Picea abies).* The latter shows growth reductions at  $Al^{3+}$  concentrations of 20-27 mg/L (Abrahamsen, 1984; Eldhuset and Göransson, 1989). These results indicate that Norway spruce may respond adversely to a further increase in exchangeable AI in the soil.

#### **Effects of Aluminium on Aquatic Biota**

Although high A1 concentrations in waters polluted by industrial sources are toxic to invertebrates and fish (Hunter *et al.,* 1980; Lamb and Bailey, 1981), it is the acidification caused by acid rain that has caused the ecological significance of Al toxicity.

Aquatic plants have so far not been demonstrated to be affected by elevated levels of A1 in acidified freshwaters (B. Rørslett, personal communication), pH and A1 affect both invertebrates and fish, and the effects are dependent not only on species but also on life history stage of the animals. In the field, the effects of A1 alone is difficult to isolate from a variety of potentially interrelated adverse factors. Especially during episodes, large variations in pH, Al-species distribution and calcium can occur (Henriksen *et al.,* 1984; Gagen and Sharpe, 1987; Lacroix and Townsend, 1987); all influence the biological response. In the laboratory, the effects of A1 *per se* can be studied at given levels of pH, Ca *etc.* The results from such studies will be used in the following, presenting only the specific Al-response.

In experiments, Al has been added either as AlCl<sub>3</sub> or  $\text{Al}_2$  (SO<sub>4</sub>)<sub>3</sub>. The chemical used might be of importance as to the specific animal response, as many experiments using  $Al<sub>2</sub>(SO<sub>4</sub>)<sub>3</sub>$  have induced an increased mucus production at the fish gills, whereas use of AlCl<sub>3</sub> or natural acidic waters have not. In the following, however, no distinction is made between experiments using the different chemicals.

Whenever a specific response is related to certain species of aluminium, *i.e.* the monomeric inorganic species termed "labile AI" (LA1), the separation technique of Driscoll (1984) with or without minor modifications, has been used.

#### **Effects of Aluminium on Invertebrates**

Acidification has generally been accompanied by declining numbers of both planktonic and benthic invertebrates (Leivestad *et al.,* 1976; Haines, 1981; Okland and Okland, 1986). However, the mechanisms by which A1 *per se can*  act as a harmful agent on these organisms are largely unknown (Herrmann, 1987a).

Hörnström et al. (1984) indicated that Al could affect the zoo-plankton community in acidified surface waters. On adding A1 to an acid stream, Hall *et al.* (1985; 1987), Raddum and Fjellheim (1987) and Ormerod *et al.* (1987) observed an increased drift of mayfly nymphs, chironomids and .dixid midges. Many of the animals related to the surface film were found dead, presumably caused by reduction of surface tension; this was indicated by a pronounced foam production (Hall *et al.* 1985; Ormerod *et al.,* 1987). A1- induced mortality on stoneflies, the isopoed *Asellus* and caddis larvae was reported by Burton and Allan (1986), who also demonstrated a reduced mortality whenever the organic content of the water was high. The counteracting effect of humic acids relative to Al-toxicity was also demonstrated by Petersen et al. (1986) on blackfly larvae.

Not all invertebrate species tested have shown high sensitivity to A1. No additional mortality due to A1 was observed on bivalves and gastropods (Mackie, 1986) or crayfish (Berrill *et al.,* 1985) in acidic waters. Appelberg (1985), however, demonstrated reduced haemolymph  $Na<sup>+</sup>$ content in crayfish exposed to acidic Al-rich waters.

At very low pH, high concentrations of A1 can have an ameliorating effect, on for example mayfly nymphs *(Heptogenia sulphurea)* (Herrmann, 1987a) and small planktonic crustacean *(Daphnia magna)* (Havas, 1985; Havas and Likens, 1985). The mechanisms involved might be the same as found for fish (see below), but in both cases the actual concentrations of A1 are much higher than normally found in acidic waters containing these organisms.

Raddum and Steigen (1981) found that the caloric content of stone flies and caddis flies from acidic rivers was lower than from more neutral rivers; this implies an increased energy consumption (metabolism) in acidic waters. Increased respiration was later demonstrated to be a response to AI, most pronounced for the most sensitive mayfly species (Herrmann and Andersson, 1986). As is the case with fish (Rosseland, 1980), pH alone seemed less important for the respiratory response.

Aluminium can also impair reproduction, shown on *Daphnia magna* (Beisinger and Christensen, 1972).

Otto and Svensson (1983) suggested ,that as in fish, A1 affects invertebrates by disturbing osmoregulation. Appelberg (1985) demonstrated a reduced haemolymph  $Na<sup>+</sup>$  content in crayfish in response to Al, and Malley and Chang (1985) showed a reduced Ca<sup>2+</sup> uptake. In *Daphnia magna*, Al reduced the  $Na<sup>+</sup>$  influx and to a lesser extent increased the outflux, thus impairing osmoregulation (Havas and Likens, 1985). The temporal reduced outflux at low pH might explain the reported beneficial effects of A1 at low pH. Witters *et al.* (1984) demonstrated reduced haemolymph Na<sup>+</sup> content in *Corixa* exposed to high Alconcentrations, and Herrmann (1987b) found that A1 caused a reduced  $Na<sup>+</sup>$  content of mayfly nymphs at low pH.

As in fish, AI acts on the respiratory organs of invertebrates, for example, the anal papillae of the phantom midge (Havas, 1986). This might explain why air-breathing invertebrates like the waterboatmen *(Corixa) are* very tolerant to acidic waters (Vangenechten *et al.,* 1979; Witters *et al.,* 1984).

Aluminium can accumulate in the bodies of invertebrates living in acidic waters (Hall and Likens, 1981; Herrmann, 1987a). Insects with aquatic larvae and nymph stages leave their "metal content" in their excuviae on emerging, thus only the water stage in their life history is metal-contaminated (Otto and Svensson, 1983). Birds such as the pied flycatcher, which lives on insects in or close to acidic lakes, have been reported to have high AIconcentrations in bone marrow and eggs indicating a food-chain transport (Nyholm, 1981). The impairment of egg hatching of these birds might therefore link the environmental catastrophy appearing in the acidic aquatic ecosystem to the terrestrial.

#### **Effects of Aluminium on Fish**

Until recently, pH alone was considered to be toxic at the egg stage with an increasing influence of AI with age after hatch (Baker and Schofield, 1980; 1982; Wood and McDonald, 1982). Leivestad *et al.* (1987), however, demonstrated that A1 reduced both the ion uptake at the eyed-egg stage and the activity of the Na-K-ATPase in the embryo.

After hatch, the main target organ for the Al-effects is the gill where the ion and gas exchange takes place. In addition to  $H^+$ , Al causes loss of plasma ions (Na<sup>+</sup>, Cl<sup>-</sup>), reduced osmolality, and increased hematocrit (Muniz and Leivestad, 1980; Rosseland, 1980; Rosseland and Skogheim, 1982; 1984; 1987; Rosseland *et al.,* 1986a, b; Fivelstad and Leivestad, 1984; Neville, 1985; Witters, 1986; Leivestad *et al.* 1987; Wood and McDonald, 1987).

In acidic waters (pH 4.6-5.3) with low levels of calcium (0.5-1.5 mg Ca/L), labile Al between 25-75  $\mu$ g/L is toxic (Henriksen *et al.,* 1984; Rosseland *et al.,* 1986a; Rosseland and Skogheim, 1987; Skogheim and Rosseland, 1986). The Al-induced ion loss reflects both an increased outflux and a decreased influx of ions (Dalziel *et al.,* 1986; 1987; Wood and McDonald, 1987). The effect on influx is probably caused by a reduced activity of enzymes as Na-K-ATPase, Mg-ATPase and carbonic anhydrase (Staurnes *et al.,* 1984; Kjartansson, 1984; Leivestad *et al.,* 1987; Reite and Staurnes, 1987). Aluminium acts specifically on

the enzymes of the gill, as neither the ATPase systems in the pseudobranch or the kidney was affected (Kjartansson, 1984). The Al-induced efflux is considered to reflect modifications on opening of the tight junctions of the paracellular channels (Wood and McDonald, 1987). The ameliorating effect of Ca on A1- and pH response (Leivestad *et al.*, 1980; Brown, 1983) is by tightening of the junctions, thereby preventing the passive loss of ions (Wood and McDonald, 1987).

Fish exposed to acidic Al-rich waters will accumulate Ai on the gill surface (Schofield, 1977; Schofield and Trojnar, 1980; Muramoto, 1981; Buergel and Soltero, 1983; Pagenkoff, 1983; Skogheim *et al.,* 1984; Neville, 1985; Karlsson-Norrgren *et al.,* 1986a, b; Harvey and McArdle, 1986; Witters *et al.,* 1987; Wood and McDonald, 1987; Jagoe *et al.,* 1987). The precipitation and accumulation is due to the negative charge of the mucus caused by sialic acid residues (McDonald, 1983). The gill also serves as an excretion organ for ammonia  $(NH<sub>4</sub> +)$ (Masoni and Payan, 1974). At low pH and high A1, the reduced blood pH (acidosis) and increased CO2 (hypercapnia) will interfere with the formation from ammonium (NH3) to ammonia, thus more is excreted as NH3. At the interface between mucus and water, the ammonium will be transformed to ammonia, changing the pH and thus enhancing precipitation of A1 at the gill surface (Wood and McDonald, 1987).

There are species and strain differences in sensitivity to low pH (Grande *et al.* 1978; Gjedrem, 1980) and A1 (Rosseland and Skogheim, 1984; 1987; Rosseland *et al.,*  1986b; Wood and McDonald, 1987). These species differences are also reflected in the accumulation rate of AI on the gills, as well as the whole body (wb) ion concentration, *i.e.* fish with the lowest wb Na (greatest loss) have the highest Al-concentration on the gill surface (Wood and McDonald, 1987). Precipitated Al-complexes can irritate the gill and cause inflammation, oedema, swelling and sometimes irradation of the secondary lamella (Schofiled, 1977; Schofield and Trojnar, 1980; Karlsson-Norrgren *et al.,* 1986a, b; Jagoe *et al.,* 1987). Also an increased number of mucus cells (Linnenbach *et al.,*  1987) and chloride cells (Jagoe *et al.,* 1987) have been observed relative to Al-accumulation onto the gills. In spite of high Al-concentration on the gill these histopathological changes are not observed when the humus content in the water is high, indicating a labile Al-dependent irritation (Karlson-Norrgren *et al.,* 1986b).

Although increased levels of AI in blood plasma have not been found (Neville, 1985; Wood and McDonald, 1987), M-accumulation in body tissue does occur (Hunter *et al.,* 1980; Muramoto, 1981; Buergel and Soltero, 1983; Haines *et al.,* 1987). In the field, such accumulation might reflect both a direct gill-dependent uptake and a food chain dependent uptake (Haines *et al.,* 1987). It was suggested by Muramoto (1981) that A1 could pass through the gill as metal-complexes in the presence of complexing ligands.

Sometimes extensive mucus-clogging of the secondary lamella has been observed (Muniz and Leivestad, 1980; Rosseland, 1980; Rosseland and Skogheim, 1984). This response is not universal, as fish dying in field in natural acidic waters at labile A1concentrations of 59-110  $\mu$ g/L have not shown excess mucus despite an Al-accumulation on the gills (Skogheim *et* a/.,1984; Rosseland *et al.,* 1986a). Adding excess aluminium as  $Al_2(SO_4)$ <sub>3</sub> to such waters (LA1 130  $\mu$ g/L) rapidly induced mucus clogging (Muniz and Leivestad, 1980; Rosseland, 1980; Rosseland and and Skogheim, 1984). The relevance of the mucus clogging might therefore be questioned with respect to natural conditions.

Both histophathological changes and increased mucus layer will serve to increase the diffusion distance for  $O<sub>2</sub>$ and CO2 between the water and blood. This can lead to a decreased oxygen tension in the arterial blood, reduced hemoglobin oxygenation and pH, and an increased blood CO2 and blood lactate (Neville, 1985; Malte, 1986; Wood and McDonald, 1987). At low pH, the increased mucus layer will reduce the rate of ion loss, thereby temporarily increasing the resistance, as observed by Baker and Schofield (1982), Hutchinson *et al.* (1987) and Wood and McDonald (1987). At such high concentrations of  $H<sup>+</sup>$  and AI, the primary cause of mortality might thus be respiratory rather than osmoregulatory failure (Rosseland, 1980; Muniz and Leivestad, 1980; Neville, 1985; Wood and McDonald, 1987).

Metabolic activity, measured as oxygen uptake, is not affected by  $H<sup>+</sup>$  alone, but increases as a response to Al in the water (Rosseland, 1980; Neville, 1985; Malte, 1986; Wood and McDonald, 1987). The increased respiratory and heart rate observed in acidic waters (Rosseland, 1980; Muniz and Leivestad, 1980; Ogilvie and Stechey, 1983; Giles *et al.,* 1984; Fivelstad and Leivestad, 1984; Neville, 1985; Malte, 1986; Leivestad *et al.,* 1987; Wood and McDonald, 1987) are not believed to cause the increased energy expenditure *per se,* as the increased metabolism rather reflects the increased activity of the intrinsic compensatory mechanism trying to restore homeostasis (Rosseland, 1980). In long-term experiments, even low concentrations of A1 have reduced growth (Sadler and Turnpenny, 1986). Hyperventilation in acidic waters is a specific response to the labile Al-concentration, as the addition of chelator such as citrate depresses hyperventilation (Leivestad *et al.,* 1987).

Prolactin and cortisol are important hormones related to osmoregulation (Potts and Flemming, 1970; Johnson, 1973); prolactin reduces ion-permeability and increases mucus production, while cortisol stimulates the onset of cellular proliferation and differentiation in the primary gill epithelium, and increases the specific activity of Na-K-ATPase. Both hormones are affected by acidic waters (Wendelaar Bonga and Balm, 1987). Plasma cortisol increases in fish exposed to low pH and high AI concentration, presumably as a response to compensate the H<sup>+</sup>/Al- response (Kjartansson, 1984). Prolactin production increases in acidic waters mainly as a response to a drop in plasma electrolytes (Wendelaar Bonga *et al.,* 1987).

Avoidance reactions to low pH waters have been observed when plasma cation concentrations have been reduced by acidic waters (Pedder and Maly, 1987). Olfaction is an important part of behavioural response and can lead to both positive and negative chemotaxes, including avoidance. Low pH alone reduces the olfactory response to aminoacids and increases the mucus layer in

the olfactory organ (Thommesen, 1975; Klaprat *et al.,*  1988). Adding A1 to the water depresses olfactory response even more and causes histopathological changes such as irradation of the microvillie, swelling and disformation of the olfactory epithelium (Klaprat *et al.,* 1988). During episodic changes in water quality related to snowmelt or heavy rain, fish are often observed gathered at the outlet of, or having migrated into, a less acid brook or stream (Muniz *et al.,* 1978; Rosseland, 1986). Chronic exposure to sublethal acidic conditions causing disturbances of the olfactory sense prior to a toxic episode, might thus reduce the chance for a fish population to find refuges and survive in their environment.

# **Effects of Aluminium on Terrestrial Animals**

When related to other metals, the toxicity of dietary AI has been regarded to be low. The intestinal absorption of orally-ingested AI salts is poor, and the small amount absorbed is almost completely excreted in urine under conditions with normal kidney function (Ganrot, 1986; Sørensen et al., 1974). However, from medical research Al is known under certain conditions to cause severe disturbances, especially in the mineral balance and nervous system, mostly in connection with renal failure. In the last decade an increasing number of human as well as animal studies have been conducted, both *in vivo* and *in vitro,* and these reveal several physiological and biochemical implications of AI (reviewed by Haug, 1984; Siegel, 1985; Ganrot, 1986; Trapp, 1986).

Environmental impacts of A1 on terrestrial wildlife, however, are poorly known. The only experimental evidence for such a connection is that of Nyholm (1981), who proposed a possible etiological role of AI in breeding impairment observed in wild passerines at some lakes in Swedish Lapland. The impairment was caused by severe eggshell defects as well as reduced clutch sizes and high incidences of mortality. Since these findings were restricted to birds nesting by the shore of a lake suspected to be acid-stressed, some kind of poisoning associated with the lake was proposed (Nyholm and Myhrberg, 1977). Extensive analyses excluded DDT, PCB and metals such as Cd, Cr, Cu, Pb and Hg to be the causal factors (Nyholm and Myhrberg, 1977). In 1981, however, the specific occurrence of AI in birds producing defective egg- shells was demonstrated semi-quantitatively by X-ray multielement microanalysis in bone marrow tissue of humeri of pied flycatchers *(Ficedula hypoleuca),* one of the most affected species (Nyholm, 1981). A possible route of transport of A1 to the birds was presented, in which insects swarming from the acidic lakes was the proximate link. Emerging insects collected from the lake contained on dry weight basis 70-1,230 mg Al/kg (Nyholm 1982). In comparison, Hall and Likens (1981) reported  $840 \pm 140$  mg A1/kg in aquatic insects in an artificially acidified stream of pH 4.

Several findings, such as successively increasing severity of the shell defects with the egg-laying sequence, very porous structure of the defective shells, and less apatite depositions in marrow cavities of limb bones (a necessary Ca storage for formation of eggshell, Simkiss (1967)), indicated Ca depletion (Nyholm, 1981).

The complex mechanism by which A1 induces osteomalecia is not yet fully understood. The presence of A1 in bone marrow associated with the reduced apatite deposition (Nyholm, 1981) seems, however, to be in agreement with the principal results from human studies as well as animal experiments. Both from *in vivo and in vitro*  studies A1 is reported to accumulate in the calcification front of bones, inhibiting calcium phosphate precipication by some physical chemical action. AI may also interfere with biochemical reactions as well as cell functions important in the mineralisation processes (reviewed by Kanis, 1981; Parkinson *et al,* 1981; Goodman, 1985; Kreuger *et al.,* 1985; Ganrot, 1986; Starkey, 1987).

From the study area of Nyholm (1981), two to three times higher blood plasma Ca have been found in affected pied flycatchers relative to birds producing normal eggshells several kilometres from the lake shore. Also the Ca concentration was similar or even somewhat higher in the shell gland tissue of affected birds compared to non-affected birds (Staumes, M. and Nyholm, N.E., in preparation). Both these findings may also indicate some direct dysfunctions in the shell formation processes.

Calmodulin is a multifunctional, Ca-dependent protein that regulates a variety of cellular reactions (Cheung, 1980). This Ca-binding protein seems to be of crucial importance in stimulation of the shell gland secretion (mediation of Ca- dependent formation of prostaglandins, and Ca-dependent binding of progesterone to its receptors), as well as the ATP-dependent transport of Ca into the shell gland lumen (Lundholm, 1987). This process involves the enzymes Ca-Mg- and Na-K-ATPase. Another enzyme important in the egg-shell formation is carbonic anhydrase (Simkiss, 1967; Simkiss and Taylor, 1971). In fish gills, A1 inhibits these enzymes (Kjartansson, 1984, Staurnes *et aL,* 1984; Leivestad *et al.,* 1987). A1 binds to calmodulin (Siegel and Haug, 1983b). As the eggshell-thinning effect of the biocide DDE mostly seems to be mediated by its inhibition of calmodulin (Lundholm, 1987), a similar role of AI might be hypothesised.

A four-month feeding period with an experimental diet supplemented with 1,000 mg AI/L as A1 sulphate, however, did not result in any effect on egg production, fertility, hatchability, or fledging success in ringed turtle-doves, *Streptopelia risoria* (Carrière et al., 1986). Compared to controls given the same diet without supplemental AI, egg permeability was decreased initially but subsequently recovered to a normal level. Plasma Ca, P, and Mg of adults were not affected. Dietary A1 (1,500 mg/L, 63 days) did not affect growth rate of juveniles although A1 tended to accumulate in the bones of sternum, but not in leg and wing bones. In treated adults, however, there was a significant accumulation also in leg and wing bones. The overall conclusion, however, was that A1 had no significant influence which could support Nyholm's (1981) hypothesis of an adverse effect of A1 on the Ca-metabolism.

Japanese quail *(Coturnix coturnixjaponicia)* has been shown to be especially sensitive to reproductive effects of Pb (Edens and Garlich, 1983). Al-citrate in drinking water

(250 and 750 mg A1/L, 37 days), however, did not cause reproductive failure in spite of elevated levels of A1 in bones was well as liver, kidney and brain (Nyholm, N.E., Gohenson, A. and Paulsson, J., in preparation).

Both in the experiments of Carrière et al. (1986) and Nyholm *et al.* (in preparation), the diet Ca content was about 1% and P content 0.5-0.9%. Carrière *et al.* (1986) stated that the lack of effect in their study could be due to higher concentrations of both Ca and P in the experimental diets compared to the feed levels which may occur in acidified areas. In rabbits, A1 ingestion lowered the inorganic P in plasma and bone (Cox *et al.* 1931). In humans, extensive use of M-containing antacida can result in a similar P-depletion which may cause osteomalacia (Spencer *et al.,* 1975; Spencer and Lender, 1979). Dietary A1 has also been reported to lower plasma P levels in poultry species (Deobald and Elvehjem, 1935; Storer and Nelson, 1968; Lipstein and Hurwitz, 1981), but Street (1942) concluded from experiments with rats that such lowering is likely to occur only when the concentration of A1 equals the concentration of P. Experiments with lower P level than A1 (dose range 2,000-25,000 mg/L as sulphate and hydroxide) in the diet of young chicks has shown to cause decreased growth and increased mortality (Deobald and Elvehjem, 1935; Storer and Nelson, 1963) as well as wing and leg weakness (Steinborn *et al.,* 1957). These effects, which Deobald and Elvehjem (1935) found most marked when dietary AI levels attained more than 50% of the P dietary level, were ameliorated by supplemental dietary P (Steinborn *et al.,* 1957; Lipstein and Hurwitz, 1981).

Thus, with respect to potential environmental effects of A1, the risk for a similar P-depletion due to restricted availability and intestine retention is obvious in both mammals and birds. In most acidified areas, Ca concentration is low, at least in the freshwaters. In association with Pb intoxication, low dietary Ca is known to increase strongly the intestinal absorption of Pb in mammals (Six and Goyer, 1970; Mahaffey *et al.,* 1973; Stowe *et al.,* 1973; Van Barneveld and Van den Hamer, 1985), as well as in birds (Carlson and Nielson, 1985). Pb can occupy the Ca-binding sites of an intestinal Ca-binding protein (Barton *et al.,* 1978), thus representing an efficient route of intestinal absorption. The basis for the increased uptake may be induction of synthesis of this protein in response to the low dietary Ca (Scheuhammer, 1987). As a result the Pb accumulation in egg-laying birds is several times that of non-laying females (Finley and Dieter, 1978) and in males (Kendall and Scanlon, 1981). In view of the binding of A1 to calmodulin (Siegel and Haug, 1983b), the possibility for a similar effect of AI should be elucidated. Moreover, both Carrière et al. (1986) in their experimental administration of Al-sulphate to ringed turtle-doves, and Nyholm *et al.* (in preparation) feeding Al-citrate to Japanese quail, reported much higher A1 absorption and accumulation in laying females than in males. This might indicate a similar Ca-deficiency induced increase in Al-absorption as was demonstrated for Pb.

Increased parathyroid activity and vitamin D level found in egglaying birds is known to enhance the absorption of several metals in chicks (Worker and Migicovsky, 1986a, b). In pregnant women there is an elevated level of PTH (Cushard *et al.,* 1972). The role of parathyroids and PTH in relation to A1 accumulation has been a matter of dispute, some claiming that PTH might increase the A1 absorption (Mayor *et al.,* 1977; 1980), while this has been questioned by others (Alfrey *et al.,*  1979; Alfrey, 1980).

Nevertheless, in view of the effects of PTH on absorption of other metals and the findings of increased A1 accumulation in laying birds, especial attention should be paid to the possibility of A1 intoxication during periods of high demands of Ca as in rapidly growing young animals. This is particularly relevant during reproduction where increased levels of PTH may mediate AI absorption and accumulation. Naturally, the risk for such an intoxication would be highest in areas with a high load of bioavailable A1 accompanied with low availability of Ca. Both conditions normally hold for acidified areas. Interestingly, secondary hyperparathyroidism provoked by chronic deficiency of Ca, Mg and causing increased absorption of toxic metals including A1, has been postulated by Yase (1977) to be the most plausible explanation to the high incidence of amyothrophic lateral sclerosis (ALS) and Parkinsonism-dementia (PD) in some indigenious populations in western Pacific. Red A1- and Fe-rich soil with very low Ca content is common to these areas. Al has been suggested to be important in the etiology since the reported AI content in spinal cord and brain as well as the pathologic changes are comparable to those from human and animal studies where A1 has been associated with disorders in the central nervous system (discussed by Ganrot, 1986). The neurotoxicity of A1, characterised by degeneration of neurons with A1 accumulation in dense tangles (neurofibrillary degeneration) and interactions with several biochemical reactions, is well documented from both human and animal studies (reviewed by Alfrey, 1978; Crapper *et al.,* 1978; 1981; Liss, 1980; Parkinson *et al.,*  1981; Kreuger *et al.,* 1985; Perl, 1985; Ganrot, 1986; Starkey, 1987).

In nature even minor behavioural abnormalities may be critical for reproduction success and survival. Chronic exposure to neurotoxic contaminants might cause changes in behavioural pattern at levels insufficient to produce increased mortality or other acute effects (Heinz *et al.,*  1983; Peakall, 1985; Donald *et al.,* 1986). Animal experiments have shown that A1 administration may be accompanied by behavioural and motor disturbances (Crapper and Dalton, 1973; Crapper *et al.,* 1973; Petit *et al.,* 1980; 1985; Commissaris *et al.,* 1982; Rabe *et al.,*  1982; Yokel, 1984; 1985; 1987; Bernuzzi *et al.,* 1986) even at AI levels that do not cause overt signs of ill health (Commissaris *et al.,* 1982). Since neurotoxicity of AI normally has been associated with slow- operation, low-dose effects, often related to ageing processes of the individual neurons as well as the whole organism (reviewed and discussed by Ganrot, 1986), it is reasonable to anticipate that longliving animals would be most vulnerable. These are normally at the top of the food chain and also most vulnerable to accumulation of other pollutants. In acidified areas the increased A1 load in freshwaters is also accompanied with higher bioavailable **concentrations of other metals (Haines** *et al.,* **1987), and elevated levels of several heavy metals are demonstrated in terrestrial animals from areas exposed to acid rain**  compared to uncontaminated areas (Frøslie et al. 1984; **1985). However, nothing is known about potential synergistic effects of AI with other metals, but contaminants interfering with the renal functions as Cd**  *(White et al. 1978; Goyer et al., 1984) and Hg (Ware et al.,* **1975), would be suspected to strengthen potential deleterious effects of A1 due to the risk for decreased excretion and thus increased accumulation.** 

**Routes for entry into and degree of accumulation of A1 along food chains have not been investigated. The findings of Nyholm (1981; 1982) indicate that increased A1 concentrations in streams and waters may cause A1 accumulation in terrestrial animals eating prey originated from contaminated water.** 

**The forest vole** *(Clethrionomus glareolus)* **is one of the small rodents which are correlated to be key organisms in the terrestrial food chain. Feeding on blueberries from an acidified area (containing 156 mg AI/L) resulted in a significant higher A1 content in bones than feeding on**  blueberries from an uncontaminated area (57 mg Al/L) (E. **Nyholm, P. Mattson and P. Slanina, in preparation). Thus, the increased concentrations of labile AI in acidified soil may enter the food chain directly through plants. Human and animal studies have shown that complexes as Al-citrate are much more readily absorbed and accumulated than the inorganic salts of A1 (Slanina** *et al.,* **1984; 1986; Yokel and McNamara, 1987). The short period of exposure of the mice (one week), along with the relatively moderately elevated levels of AI in the blueberries, may indicate that AI in plants is bound to similar easily-absorbed complexes which strongly enhance the bioavailability of A1. The potential toxicity of such easily-absorbed complexes is seen from an experiment with one week old chicks of wood grouse** *(Tetrao urogallus),* **where addition of Al-citrate to commercial bird feed to give 4,000 mg AI/L strongly increased the mortality compared to those given uncontaminted feed; none survived two weeks exposure (M. Staurnes and T.K. Spidso, in preparation). The same dose of AI caused accumulation in kidney and liver of five week old chickens, but no mortality. Similarly, Nyholm observed especially high mortality just after hatching in the populations of passerines with impaired breeding in**  Swedish Lapland (N.E. Nyholm, personal communication). **Thus, at least in birds, the greatest risk for acute intoxication seems to be during the earliest life stages.** 

**Future studies, however, will show if the increased concentration of AI associated with acidification, either as labile AI or organic-complexed represents a potential risk for birds and mammals as well as other terrestrial animals. The degree of and routes for bioaccumulation, differences in species and life-stage sensitivity, occurrence and toxicity of different AI compounds, potential synergistic effects with other contaminants, and potential interactions because of restricted availability of substances as Ca and P will all be key questions in future studies. A comparative study of wildlife in regions with a naturally high concentation of A1, as those mentioned areas in western Pacific, where occurrence of ALS and PD are high, could possibly**  **contribute to a better understanding of potential environmental hazards of A1 as well as adaptation mechanisms.** 

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