APPENDIX J

SENSITIVITY OF AQUATIC BIOTA TO ACIDIFICATION

Extensive information is available on the effects of acidification on aquatic communities. Whole system experiments, mesocosm experiments, and field surveys have demonstrated major shifts in species composition and decreases in species richness with increasing acidity. The range of sensitivity to acidification varies among fish species, and to a greater extent among invertebrate species. Some sensitive species are lost at even moderate pH levels. In lakes, some important zooplankton predators are affected at pH 5.6-5.9. Some sensitive mayflies and fish (e.g. *Baetis lapponicus*, fathead minnow) are lost at pH 5.6- 6.0 (Baker and Christensen 1991). Toxic mechanisms are well established for fish, and for invertebrates to a lesser degree (Baker et al. 1991).

Direct hydrogen ion toxicity occurs in all aquatic taxa tested, but sensitivity varies greatly among species and higher taxa (and life history stages), which accounts for the graded response in species loss from communities. Ionoregulatory failure caused by acid stress leads to death in invertebrates as well as fish.

Sensitivity of Fish to Acidification

Mechanisms of acid toxicity to fish

Both elevated concentrations of hydrogen ion (measured by low pH) and aluminum (mobilized by low pH) are directly toxic to fish. Aluminum is the most abundant metal on the earth's surface, and the third most abundant element. It is non-toxic and insoluble under acid-neutral conditions, but very toxic to fish and some other aquatic species when it is found in solution. The solubility of aluminum increases dramatically as pH falls below 5.6; its maximum toxicity occurs at about pH 5.0. The deposition of acids results in the release of aluminum from soils, which is carried in solution to streams and lakes. Both the aluminum and the hydrogen ion (derived from sulfuric and nitric acids) are toxic to fish, but in most streams and lakes the aluminum is the primary lethal agent. The site of the toxic action of both hydrogen ion and aluminum is the fish gill. The gill is a complex organ responsible for oxygen and carbon dioxide exchange, as well as maintaining the proper salt and water balance in the fish's body. It is this latter function which is always compromised by acid and aluminum stress; respiration is also compromised at higher concentrations of aluminum (Bulger et al. 1993).

Freshwater fish maintain salt concentrations in the blood at much higher concentrations than the water in which they swim, so fish constantly lose a small amount of sodium and chloride from the blood by passive diffusion across the thin skin of the gills. The lost sodium and chloride are replaced by an energy-requiring process (active transport) using biochemical "pumps" in the gill membranes which transport sodium and chloride from low concentration in the external stream water to higher concentration in the blood.

Aluminum and hydrogen ion poison the biochemical pumps which transport sodium and chloride into the body. They also weaken the junctions between gill cells, making them leak more sodium and chloride than they would otherwise. The rapid loss without replacement of sodium and chloride produces a cascade of negative physiological effects in the fish's body. When sodium and/or chloride concentrations fall more than 30% below normal, death occurs within hours. Fish may recover from less severe stress, although at substantial metabolic cost.

The proximal cause of death is ionic dilution of the blood plasma. This causes blood and body fluid disturbances which ultimately kill the fish through circulatory collapse. Under normal conditions, plasma ionic (electrolyte) concentrations and body cell ionic concentrations are in equilibrium. Under acute acid stress, ions are lost more rapidly from the blood plasma than from blood and muscle cells; as a result, there is an osmotically-driven shift of water to the cells from the plasma. Blood plasma volume may drop as much as 30%; at the same time, the red cells swell due to the osmotically-driven shift of water from the plasma; the result is a doubling of blood viscosity. The heart is unable to circulate this much thicker blood at a rate sufficient to supply oxygen to body tissues, including the heart itself, so the fish dies of circulatory collapse secondary to ionic imbalance (Wood 1989).

Differential fish species sensitivity

Although there are known differences among fish species in acid sensitivity, experimentally determined acid sensitivities are only available for a minority of freshwater fish species. For example, of 28 species of fish found in Shenandoah National Park, the critical pH is known for only nine. Baker and Christensen (1991) reported critical pH values for 25 species of fish. The critical pH is the threshold for significant demonstrated adverse effects on populations. The range of pH values represents the authors' estimate of the uncertainty of this threshold. The range of response within species depends on differences in sensitivity among life stages, and on different exposure concentrations of calcium and aluminum. These ranges are based on multiple studies for each species shown in Table J-1. To cite a few examples, blacknose dace is regarded

Table J-1. Fish Species of Shenandoah National Park and the known critical pH thresholds for serious adverse impacts (Source: Bulger et al. 1999).

Common Name	Latin Name	Family	"Critical pH" Thresholds**
American Eel	*Anguilla rostrata*	Anguillidae	
Mtn. Redbelly Dace	*Phoxinus oreas*	Cyprinidae	
Rosyside Dace	*Clinostomus funduloides*	Cyprinidae	
Longnose Dace	*Rhinichthys cataractae*	Cyprinidae	
Blacknose Dace	*Rhinichthys atratulus*	Cyprinidae	5.6 to 6.2
Central Stoneroller	*Campostoma anomalum*	Cyprinidae	
Fallfish	*Semotilus corporalis*	Cyprinidae	
Creek Chub	*Semotilus atromaculatus*	Cyprinidae	5.0 to 5.4
Cutlips Minnow	*Exoglossum maxillingua*	Cyprinidae	
River Chub	*Nocomis micropogon*	Cyprinidae	
Bluehead Chub	*Nocomis leptocephalus*	Cyprinidae	
Common Shiner	*Luxilus cornutus*	Cyprinidae	5.4 to 6.0
Northern Hogsucker	*Hypentelium nigricans*	Catostomidae	
Torrent Sucker	*Thoburnia rhothocea*	Catostomidae	
White Sucker	*Catastomus commersoni*	Catostomidae	4.7 to 5.2
Margined Madtom	*Noturus insignis*	Ictaluridae	
Brook Trout	*Salvelinus fontinalis*	Salmonidae	4.7 to 5.2
Brown Trout	*Salmo trutta*	Salmonidae	4.8 to 5.4
Tiger Trout*	*Salmo X Salvelinus*	Salmonidae	
Rainbow Trout	*Oncorhynchus mykiss*	Salmonidae	4.9 to 5.6
Mottled Sculpin	*Cottus bairdi*	Cottidae	
Rock Bass	*Ambloplites rupestris*	Centrarchidae	4.7 to 5.2
Smallmouth Bass	*Micropterus dolomieui*	Centrarchidae	5.0 to 5.5
Largemouth Bass	*Micropterus salmoides*	Centrarchidae	
Redbreast Sunfish	*Lepomis auritus*	Centrarchidae	
Pumpkinseed	*Lepomis gibbosus*	Centrarchidae	
Johnny Darter	*Etheostoma nigrum*	Percidae	
Tesselated Darter	*Etheostoma olmstedi*	Percidae	
Fantail Darter	*Etheostoma flabellare*	Percidae	
Greenside Darter*	*Etheostoma blennioides*	Percidae	

* Tiger trout (a hybrid) and greenside darter are rare in the park; 10 or fewer individuals have been documented.

** Thresholds taken from Baker and Christensen (1991) as very sensitive to acid stress, because population loss due to acidification has been documented in this species at pH values as high as 6.1; in field bioassays, embryo mortality has been attributed to acid stress at pH values as high as 5.9. Embryo mortality has occurred in common shiner at pH values as high as 6.0. Although the critical pH range for rainbow trout is recorded as 5.6, adult and juvenile mortality have occurred at pH values as high as 5.9. Brown trout population loss has occurred over the range of 4.8-6.0, and brook trout fry mortality has occurred over the range of 4.8-5.9 (Baker and Christensen 1991). Relative sensitivities can be suggested by regional surveys as well. For example, about half of the 53 fish species found in Adirondack waters never occur at pH values below 6.0 (Kretzer et al. 1989, Driscoll et al. 2001a); for those species whose acid tolerances are unknown, it is likely that acid sensitivity is responsible for at least some of these absences.

It is the difference in acid tolerance among species that produces a gradual decline in species richness (the total number of fish species in a lake or stream) as acidification progresses, as the most sensitive species are lost first. Southern Blue Ridge streams can become too acid even for brook trout, as evidenced by the absence of the species from streams with mean pH < 5.0 in Great Smoky Mountains National Park (Elwood et al. 1991).

Species richness, biomass, density, and condition.

A direct outcome of fish population loss as a result of acidification is a decline in species richness. This appears to be a highly predictable outcome of regional acidification, although the pattern and rate of species loss varies from region to region. Baker et al. (1990) discussed 10 selected studies which documented this phenomenon, with sample sizes ranging from 12 to nearly 3000 lakes and streams analyzed per study. An excellent example occurs in the Adirondacks. Fully 346 of 1469 lakes surveyed supported no fish at all. These lakes were significantly lower in pH, dissolved calcium, and ANC than lakes hosting one or more species of fish. Among lakes with fish, there is an unambiguous relationship between the number of species and lake pH (Kretzer et al. 1989, Driscoll et al. 2001b).

Relatively less is known about changes in fish biomass, density and condition (robustness of individual fish) which occur in the course of acidification. Loss of sensitive individuals within species (including early life stages) may reduce competition for food among the survivors, resulting in better growth rates, survival, or condition. Similarly, competitive release may result from the loss of a sensitive species, with positive effects on the density, growth, or survival of competitor population(s) of other species (Baker et al. 1990). In some cases where acidification continued, positive effects on size of surviving fish were shortly followed by extirpation (Bulger et al. 1993).

Acidification effects on fish in the SAMI region

Shenandoah National Park, Virginia

A recent three-year study on stream acidification in Shenandoah National Park (SHEN) demonstrated negative effects on fish from both chronic and episodic acidification (Bulger et al. 1999). Biological differences in low- ANC versus high-ANC streams included species richness, population density, condition factor, age, size, and field bioassay survival. Of particular note is that both episodic and chronic mortality occurred in young brook trout exposed in a low- ANC stream, but not in a high-ANC stream (MacAvoy and Bulger 1995), and that blacknose dace in low-ANC streams were in poor condition relative to blacknose dace in higher-ANC streams (Dennis et al. 1995, Dennis and Bulger 1995).

Acidification has been shown to reduce fish species richness in many regions by eliminating sensitive species initially, followed by more tolerant species as acidification proceeds (Baker et al. 1990). A statistically robust relationship between acid-base status of stream water and fish species richness has been shown in the SAMI region as well. As an element of the Shenandoah National Park:Fish in Sensitive Habitats (FISH) project (Bulger et al. 1999), numbers of fish species were compared among 13 SHEN streams spanning a range of pH/ANC conditions. There was a highly significant ($p<0.0001$) relationship between stream acid-base status (during the seven-year period of record) and fish species richness among the 13 streams, such that the streams having the lowest ANC hosted the fewest species (Figure J-1). Although the number of streams in the study was small, the results were consistent with other studies (Baker et al. 1990); this was the first, however, to provide a statistically robust analysis among multiple streams in the southeastern US.

As another component of the FISH project (Bulger et al. 1999), condition factor (a measure of robustness in individual fish) was compared in populations of blacknose dace in 11 streams spanning a range of pH/ANC conditions in the park. Figure J-2 shows the highly significant relationship between mean stream pH and condition factor in blacknose dace. Note that the 4

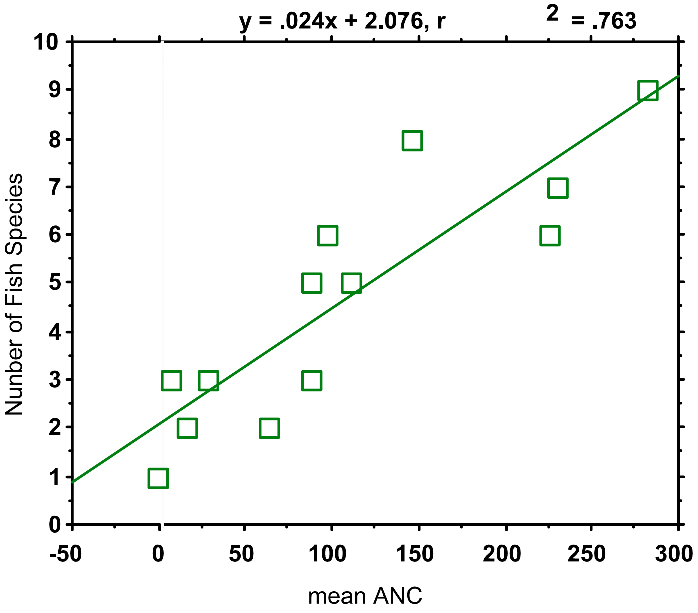

Figure J-1. Number of fish species among 13 streams in Shenandoah National Park. Values of Acid Neutralizing Capacity (ANC) are means based on quarterly measurements, 1987-94. The regression analysis showed a highly significant relationship (p = 0.0001) between mean stream ANC and species number, such that acidified streams host fewer fish species than circumneutral streams.

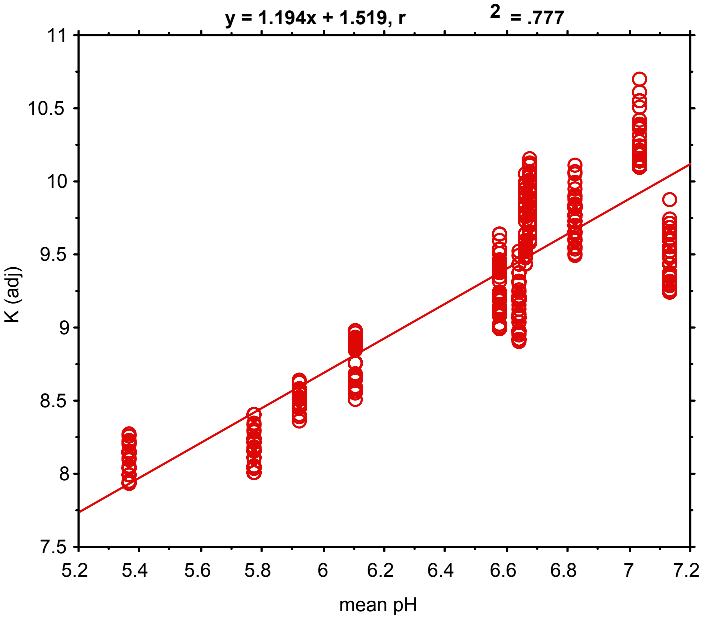

Figure J-2. Condition factor (K), a measure of body size in blacknose dace (*Rhinichthys atratulus*) among 11 populations (n = 442) in Shenandoah National Park. Values of pH are means based on quarterly measurements, 1991-94; K was measured in 1994. The regression analysis showed a highly significant relationship (p = 0.0001) between mean stream pH and body size, such that fish from acidified streams are smaller than fish from circumneutral streams.

populations represented on the left side of the figure all had mean pH values within or below the range of critical pH values for the species, at which negative populations effects are likely (Baker and Christensen 1991). That poor condition is related to population survival is suggested by the extirpation in 1997 of the blacknose dace population from the stream with the lowest pH and ANC (J. Atkinson, pers. comm.; Figure J-2).

The results of the condition factor comparisons among the 11 streams indicated that the mean length-adjusted condition factor of fish from the stream with the lowest pH and ANC was about 20% lower than that of the fish in best condition. No previous studies have reported changes in condition factor of blacknose dace during acidification. However, dramatic changes were associated with a 12% reduction in lake trout condition factor in a long-term acidification study (Schindler 1987). Comparisons with the work of Schofield and Driscoll (1987) and Kretser et al. (1989) suggest that pH in the low-pH Shenandoah National Park streams is near or below the limits of occurrence for blacknose dace populations in the Adirondack region.

Smaller body size could result from direct toxicity (e.g., elevated energy use to compensate for sublethal ionoregulatory stress) or from reduced access to food or lower food quality (Baker et al. 1990). Primary productivity is low in headwater streams and lower still in softwater headwaters, which are more likely to be acidified. Production of invertebrates is likely to be low in such streams as well (Wallace et al. 1992). Therefore, lower food availability cannot be ruled out as a potential contributor to lowered condition in Shenandoah National Park blacknose dace populations. Nevertheless, reduced growth rates have been attributed to acid stress in a number of fish species, including Atlantic salmon, chinook salmon, lake trout, rainbow trout, brook trout, brown trout, and Arctic char. Furthermore, the blacknose dace population in poorest condition in the park occurred in a stream with a mean pH below the minimum recorded for blacknose dace populations in Vermont, New Hampshire, Maine and New York (Baker et al. 1990). The four blacknose dace populations in poorest condition occur in streams at or below the critical pH for the species where adverse effects due to acidification are likely to be detectable at the population level (Baker et al 1990). Consequently, acid stress is probably at least partly responsible for the lower condition of blacknose dace populations in Shenandoah National Park, though lower food availability, either resulting from the nature of softwater streams, or exacerbated by acidification, cannot be ruled out.

It is possible that smaller body size in blacknose dace is the result of energy transfer from somatic growth to physiological maintenance, in response to chronic sublethal acidification stress. It is well known that chronic sublethal stress reduces growth in fish, as well as reproductive success (Wedemeyer et al. 1990). Chronic sublethal stress caused by pH levels up to 6.0 may have serious effects on wild trout populations (Kelso et al. 1986). There is an energetic cost in maintaining physiological homeostasis; the calories used to respond to stress are a part of the total energy budget which is unavailable for other functions, such as growth (Schreck 1981, 1982).

The energy costs to fish for active iono-osmoregulation can be substantial (Farmer and Beamish 1969, Bulger 1986). The concentrations of serum electrolytes (such as sodium and chloride) are many times higher (often 100-fold higher) in fish blood than in the freshwaters they live in. The active uptake of these ions occurs at the gills. Because of the steep gradient in sodium and chloride concentration between the blood and freshwater, there is constant diffusional loss of these ions, which must be replaced by energy-requiring active transport. Low pH increases the rate of passive loss of blood electrolytes (especially sodium and chloride); and aluminum elevates losses of sodium and chloride above the levels due to acid stress alone (Wood 1989). Since dace in an acidified stream maintain whole-body sodium at levels similar to dace in a high-ANC stream (Dennis and Bulger 1995), despite probable higher gill losses of electrolytes due to acid/aluminum stress, then the homeostatic mechanisms at the gill responsible for maintaining blood electrolyte levels must work harder and use more energy to maintain these levels.

An additional component of the FISH project used multiple bioassays over three years in one of the low ANC streams to determine the effect of stream baseflow and acid episode stream chemistry on the survival of brook trout eggs and fry (MacAvoy and Bulger 1995). Simultaneous bioassays took place in mid- and higher-ANC reference streams. Acid episodes (with associated low pH and elevated aluminum concentrations, and high streamwater discharge) induced rapid mortality in the low-ANC stream, while the test fish in the higher-ANC stream, experiencing only the high streamwater discharge, survived (Bulger et al. 1999).

St. Marys River, Virginia

St. Marys River occurs in the St. Marys Wilderness, George Washington and Jefferson National Forests in Augusta County, Virginia. In Virginia, records of stream pH and species richness are longest for St. Marys River. Fish abundance and diversity have been altered dramatically as a result of stream acidification (Bugas et al. 1999). In particular, the extirpation of rainbow trout and sharp decline in abundance of blacknose dace are symptomatic of acidified waters. Stream pH values in the St. Marys River in 1938-1976 were 6.7-7.0. The stream acidified over the next decade, such that pH values were 5.1-5.7 in 1989-1997. Fourteen fish species have been collected in St. Marys River since 1976; only four remain (1998). Rosyside dace and torrent sucker were last present in 1996; Johnny darter and brown trout were last present in 1994. Rainbow trout and longnose dace were last present in 1992; bluehead chub and smallmouth bass were last present in 1990 and 1988, respectively; white sucker and central stoneroller were last present in 1986. Of the four remaining species, three (blacknose dace, fantail darter, and mottled sculpin) have declined in density and/or biomass; the fourth remaining species is brook trout, the region's most acid tolerant species; this population has fluctuated, and reproductive success has been sporadic. Blacknose dace, once abundant throughout the river, remain only in the lowest station of the stream, which has the highest pH, and at such low numbers (five individuals in 1998) that they might be strays from downstream. For some of the species (smallmouth bass, white sucker, the three trout, and blacknose dace) the critical pH is known, and their decline and/or extirpation, given the pH of the river, is not surprising. Based on trend analysis over the period 1987-1997, St. Marys River is continuing to acidify (Webb and Deviney 1999).

After much public debate and discussion, the Forest Service has decided to lime St. Marys River, to prevent further biodiversity loss in this wilderness area. Elsewhere, following liming, if water quality improves to levels not toxic to fish, surviving remnant fish populations often successfully reproduce and increase in abundance; populations of acid-sensitive species may be stocked successfully. Growth rates of restocked fish are often temporarily higher than in non-acidified waters, suggesting that food availability played a minor role in fish decline (Baker et al. 1990). Although there are no records of restoration of an entire fish community to pre-acidification conditions, clear improvement is typical (Driscoll et al. 2001a). Given maintenance of improved acid-base status, improvement of the fish community in St. Marys River is likely.

Sensitivity of Invertebrates to Acidification

As in fish, invertebrates experience a net loss of sodium and chloride with acidification, which in some cases has been shown to result from both increased efflux and decreased influx of ions. In crustaceans and mollusks, salt and water balance stress is compounded by failure to take up calcium for shell maintenance. Also as in fish, low calcium levels exacerbate acid stress, but unlike fish, aluminum toxicity is less clear and ubiquitous, and macroinvertebrates show heterogeneous responses to aluminum and heavy metal exposure (Baker et al. 1990).

Species richness of benthic invertebrates is positively related to stream pH regionally, especially among aquatic insects such as mayflies, stoneflies, and caddisflies (Elwood et al. 1991). Among invertebrates, differences in acid sensitivity appear to span a wider range than in fish. In invertebrates as well as fish, life stages within species also differ in sensitivity. Molting in crayfish, for example, appears to be an especially acid-sensitive period, and among aquatic insects, early nymphal stages and emergence may be particularly sensitive stages (Baker et al. 1990). Acidity has dramatic effects on snail production in high-gradient SA streams. Gastropod production ranged from 1.8 to 12.5 gDW/m^2/year, with lowest pH sections having the lowest production; in general gastropods are rare or absent from streams draining crystalline bedrock, whereas they may comprise up to 95% of the invertebrate biomass in calcium-rich dolomite catchments (Wallace et al. 1992). Mussel as well as snail species require calcium for shell and carapace integrity; surface waters susceptible to acidification have low calcium levels. It is clear, however, that waters with low calcium which have not acidified have higher diversity than low-calcium waters which have acidified.

Relatively insensitive species appear to include some dragonflies, some mayflies, some caddisflies and stoneflies (Odonata, Ephemeroptera, Trichoptera, Plecoptera), whereas other stoneflies, caddisflies and mayflies are very sensitive (Baker et. al. 1990). In North American streams, acid-sensitive species typically include some mayflies (Ephemeroptera, e.g. *Baetis* spp., *Heptagenia* spp., *Ephemerella* spp., *Epeorus* spp., *Stenonema* spp.,). *Ephemerella*, for example, appears to be absent from steams with pH 5.5 or less. Other mayflies, such as *Leptophlebia*, and the stoneflies *Leuctra* and *Isoperla* appear tolerant of low pH, and therefore tend to dominate at low-pH sites. All of these insects are well represented in the SAMI region (Baker et. al. 1990, Elwood et al. 1991).

Synoptic surveys comparing circumneutral waters with waters that have pH 6 or less have demonstrated, without exception, that some populations of mayflies, amphipods, snails, and clams are lost with acidification. Within many taxonomic groups, there is a sharp drop in diversity between pH 6 and pH 5 (Baker et. al. 1990). Evidence comes from many studies, including many geographic regions (North America, Europe, and New Zealand). There is no doubt that decrease of one pH unit or more will result in the loss of sensitive species.

Although there are relatively few experimental studies of the effects of acidification on invertebrates, they confirm the acid sensitivity of species that tend to be absent from acidified waters. Short-term (hours to days) experimental acidification of streams consistently results in loss of sensitive invertebrates due to rapid drift responses (drift is passive downstream movement of displaced organisms). For example, experimental acidification of a third order stream in New Hampshire resulted in losses of the mayflies *Ephemerella* and *Epeorus*, while more tolerant forms, such as the stonefly *Leuctra*, did not respond; emergence of some mayflies, stoneflies and true flies (*Diptera*) decreased. Experiments in streamside channels, using replicated acidification treatments, have yielded similar results. Experimental acidification with a small pH drop (0.1 unit, or a doubling of hydrogen ion concentration) also significantly increased drift, suggesting that only slight depression in pH can have detrimental effects on invertebrate communities. The taxa that increase in drift after experimental acidification are the same as those that disappear from streams following long-term acidification (Baker et. al. 1990).

Synoptic surveys of invertebrates have also revealed negative correlations between acid-base status and biotic indicators such as species richness, species diversity, functional group diversity, and sometimes total biomass and density. Indices which only count species or individuals, such as measures of density, diversity and richness, may mask early acidification effects when acid-tolerant species replace acid-sensitive species. For a time, the community may be much altered, while species number or total density changes little (Baker et al. 1990).

Changes in species richness are, however, often very dramatic under acidification, especially when comparisons are made within individual taxonomic groups, such as orders of insects. Studies in Ontario and New York have shown dramatic differences in the numbers of stoneflies, caddisflies and or mayflies in acidified versus circumneutral streams. In Ontario, higher pH streams (5.3-6.7) had five genera of mayflies, five stonefly genera, and fifteen caddisfly genera, whereas acid (4.3-4.8) streams had one or no mayfly genera, only two stonefly genera, and only five genera of caddisflies. Results were similar over the same pH range in the Adirondacks. In a study of Pennsylvania streams, the index of species diversity was 50% lower in a low pH stream (pH 4.6-6.0) than in a high pH stream (pH 6.1-7.4; Baker et al. 1990).

Most surveys of lakes and streams have revealed low total biomass of benthic invertebrates at low pH. Reduced growth rates of acid stressed populations of invertebrates have been documented; these could result from direct toxicity (e.g., elevated energy use to compensate for sublethal ionoregulatory stress) or from reduced access to food or lower food quality (Baker et al. 1990).

References

Baker, J.P., and S.W. Christensen. 1991. Effects of Acidification on Biological Communities. In: D.F. Charles, ed. Acidic Deposition and Aquatic Ecosystems. Springer-Verlag, New York, pp. 83-106.

Bugas, P. E., Jr., L. O. Mohn, and J. W.. Kauffman. 1999. Impacts of Acid Deposition on Fish Populations in St. Marys River, Augusta County, VA. Banisteria 13:191-200.

Bulger, A.J. 1986. Coincident peaks in serum osmolality and heat-tolerance rhythms in seawater-acclimated killifish (*Fundulus heteroclitus*). Physiol. Zool. 59(2): 169-174.

Bulger, A. J., B. J. Cosby, C. A. Dolloff, K. N. Eshleman, J. R. Webb, and J. N. Galloway. 1999. The "Shenandoah National Park: Fish in Sensitive Habitats (SNP: FISH)" Project Final Report. An Integrated Assessment of Fish Community Responses to Stream Acidification. National Park Service. 570 pages plus interactive computer model.

Bulger, A.J., L. Lien, B.J. Cosby and A. Henriksen. 1993. Trout status and chemistry from the Norwegian thousand lake survey: statistical analysis. Canadian Journal of Fisheries and Aquatic Sciences. 50(3): 575-585.

Dennis, T.E. and A.J. Bulger. 1995. Condition factor and whole-body sodium concentration in a freshwater fish: evidence of acidification stress and possible ionoregulatory over-compensation. Water Air Soil Pollut. 85:377-382.

Dennis, T.E., S.E. MacAvoy, M.B. Steg, and A.J. Bulger. 1995. The association of water chemistry variables and fish condition in streams of Shenandoah National Park (USA). Water Air Soil Pollut. 85:365-370.

Driscoll, C. T., G. B. Lawrence, A. J. Bulger, T. J. Butler, C. C. Cronan, C. Eager, K. L. Lambert, G. E. Likens, J. L. Stoddard, and K. C. Weathers. 2001a. Acidic Deposition in the

Northeastern US: sources and inputs, ecosystem effects, and management strategies. Bioscience 51:180-198.

Driscoll, C. T., G. B. Lawrence, A. J. Bulger, T. J. Butler, C. C. Cronan, C. Eager, K. L. Lambert, G. E. Likens, J. L. Stoddard, and K. C. Weathers. 2001b. Acid Rain Revisited: advances in scientific understanding since the passage of the 1970 and 1990 Clean Air Act Amendments. Hubbard Brook Research Foundation. Science Links Publication. Vol. 1, no. 1.

Elwood, J.W., M.J. Sale, P.R. Kaufmann, and G.F. Cada. 1991. The Southern Blue Ridge Province, pp. 319-364. In: Charles, D.F. (Ed.). Acidic Deposition and Aquatic Ecosystems. Regional Case Studies. Springer-Verlag, New York.

Farmer, G.J. and F.W.H. Beamish. 1969. J. Fish Res. Bd. Can.26:2897-2821.
Fenneman, N.M. 1938. Physiography of the Eastern United States. McGraw-Hill Book Co., Inc., New York.

Kretser, W.A., J. Gallagher, and J. Nicolette. 1989. Adirondack Lakes Study 1984-1987: an evaluation of fish communities and water chemistry. Adirondack Lake Survey Corporation, Ray Brook, NY, USA.

MacAvoy, S.E. and A.J. Bulger. 1995. Survival of brook trout (*Salvelinus fontinalis*) embryos and fry in streams of different acid sensitivity in Shenandoah National Park, USA. Water Air Soil Pollut. 85:439-444.

Schofield, C.L., and C.T. Driscoll. 1987. Fish species distribution in relation to water quality gradients in the North Branch of the Moose River Basin. Biogeochem. 3:63-85.

Schreck, C.B. 1982. Stress and compensation in teleostean fishes: response to social and physical factors. In A.D. Pickering, editor. Stress and fish. Academic Press, London, pp. 295-321.

Schreck, C.B. 1981. Stress and rearing of salmonids. Aquaculture 28: 241-249.

Wallace, B., J. R. Webster and R. L. Lowe. 1992. High-Gradient Streams of the Appalachians. In Biodiversity of the Southeastern United States Aquatic Communities. Hackney, C. T., S. M. Adams and W. H. Martin, eds. John Wiley and Sons: New York. 779 pp.

Webb, J. R. and F. A. Deviney. 1999. The Acid-Base Status of the St. Marys River: Trout Stream Sensitivity Study Results. Banisteria 13:171-182.

Wedemeyer, G. A., B. A. Barton, and D. J. MeLeay. 1990. Stress and acclimation. Pages 451-490 in C. B. Schreck and P. B. Moyle, editors. Methods for fish biology. American Fisheries Society, Bethesda, Maryland.

Wood, C. M. 1989. The physiological problems of fish in acid waters. P. 125-152 in: Morris, R., E.W. Taylor, D.J. A. Brown, and J. A. Brown (eds.), Acid Toxicity and Aquatic Animals. Cambridge University Press, Cambridge.