**Assessing the Effects of the Clean Air Act Amendments of 1990 on Ecological Resources: Updated Literature Review and Terrestrial Case Study Approach**

*A White Paper from the Second Section 812 Prospective Project Team*
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## SECTION 1: LITERATURE REVIEW

### INTRODUCTION

Appendix E of *The Benefits and Costs of the Clean Air Act 1990 to 2010* (EPA 1999) reviewed available information on the ecological effects of criteria pollutants and hazardous air pollutants regulated under the 1990 Clean Air Act Amendments. This while paper expands that review, updating it to reflect research and information that has become available in recent years. In addition, in the second section of this paper, we outline an approach for assessing the ecological effects of the CAAA in a specific terrestrial ecosystem, the Adirondacks region of New York State.

Like the original, this literature review uses a hierarchical framework of biological organization to describe effects of air pollutants on environmental endpoints. We focus on acid deposition, nitrogen deposition, mercury, and tropospheric ozone because these four pollutants continue to be the best studied. We have also expanded somewhat upon the discussion of dioxins.

To update the review, we identified relevant literature generated from 1998 onwards. Although this time period is limited, the number of potentially relevant articles is still large, and it was not possible to identify and review all relevant items without setting some limits. To ensure that the updated review reflects the current state of science, we focused our initial efforts on obtaining review articles. We supplemented these with selected literature identified through more focused searches and/or items cited in the bibliographies of other articles.

Although EPA's focus on a clean environment has long included protection of both ecosystem health and human health, many past analyses, particularly economic analyses, have focused on human health benefits of pollution control. Ecological benefits, by comparison, have not always been as well-represented, for a variety of reasons:

- Ecological impacts have not been researched to the same extent as human health effects.
- Ecological impacts are not well understood because the potential range of endpoints to evaluate is vast. Human health studies focus on a single species and, often, a well-defined effect, while ecological impact studies could conceivably focus on any number of species, interactions between species and/or their environment, or ecological processes and flows.

- Ecological impacts are challenging to study, often requiring many different types of studies (laboratory research, field studies, and/or modeling) to generate the lines of evidence needed to understand the relationships between pollutants and endpoints. The high variability in ecosystems and potential for confounding (or unidentified) factors to influence results can make drawing clear conclusions especially difficult.

Nonetheless, within the last few decades air pollution started to receive attention for not only affecting human health but also its dramatic injuries to ecosystems. Increased public awareness and research results have led to the development of air pollution research as an important branch of applied biological sciences. Numerous scientific studies have revealed adverse effects of air pollution on natural systems.

This analysis attempts to incrementally expand the base of information that can be used to assess affects to ecosystems associated with air pollution. More particularly, the goal of this review to provide a broad characterization of the range of effects of major air pollutants on environmental endpoints. In most cases, we rely on published, peer-reviewed literature to establish the validity of the methods and data applied.

The remainder of this document is comprised of six major sections as follows:

- **Overview of ecological impacts.** This section introduces the process used to select the pollutants that are the focus of this review and presents the general framework used to categorize the impacts of these pollutants at various levels of biological organization.

- **Acidification associated with airborne nitrogen and sulfur deposition**. Acidification is perhaps the best studied effect of atmospheric pollutant deposition. Acidification of aquatic ecosystems has been shown to cause direct toxic effects on sensitive aquatic organisms. Chronic acidification of terrestrial ecosystems can also indirectly injure vegetation by causing nutrient deficiencies in soils and aluminum mobilization.

- **Eutrophication associated with airborne nitrogen deposition**. Deposition of nitrogen can stimulate nitrogen-uptake by plants and microorganisms and increase biological productivity and growth (*i.e.*, eutrophication). Moderate levels of nitrogen input can have a "fertilizing" effect, similar to the application of nitrogen fertilizer frequently used in timber production or agriculture. In the long run, however, chronic deposition of nitrogen adversely affects biogeochemical cycles of watersheds (*i.e.*, nitrogen saturation), and can have negative effects on the biodiversity of unmanaged terrestrial and aquatic ecosystems, especially coastal estuaries.

- **Impacts to wildlife associated with hazardous air pollutant deposition, particularly mercury and dioxins.** Like nitrogen- or sulfur-containing atmospheric pollutants, mercury is conserved in ecosystems. Atmospheric deposition of mercury and its subsequent movement in ecosystems results in the transfer of mercury to the food chain. In many ecosystems, mercury in the form of methylmercury accumulates up food webs, with increasing concentrations found in animals at higher levels of the food chain, an effect referred to as "biomagnification." This is of concern because methylmercury is a potent neurotoxicant in many forms of wildlife. Dioxins have been associated with a wide range of impacts to vertebrates, including fish, birds, and mammals. Most toxic effects of dioxin are mediated through interactions with the aryl hydrocarbon receptor.

- **Impacts to vegetation associated with ozone exposure**. The ecological significance of ozone lies in its direct or indirect toxicity to biota. Injuries caused by ozone are mainly related to inhibitions of essential physiological functions of plants and subsequent reductions in biomass production (reduced growth). These injuries can cause stand-level forest decline in sensitive ecosystems.
- **Summary of ecological impacts from CAAA-regulated air pollutants.** Overviews of ecological effects are presented in tabular form, and major conclusions are drawn.

## OVERVIEW OF THE ECOLOGICAL IMPACTS OF AIR POLLUTANTS REGULATED BY THE CAAA

Our review describes the impacts of air pollutants at various levels of biological organization. We identify single pollutant environmental effects and, where possible, the synergistic impacts of ecosystem exposure to multiple air pollutants. Although a wide variety of complex effects are described or hypothesized in the literature, for the purposes of this analysis we have limited the scope of our review according to the following criteria:

- Pollutants regulated by the CAAA (*i.e.*, criteria and hazardous air pollutants);
- Known effects of pollutants on natural systems as documented in peer-reviewed literature; and
- Pollutants present in the atmosphere in sufficient amounts after 1970 to cause significant damages to natural systems.

We note that a number of studies have evaluated impacts of criteria pollutants on specific endpoints and failed to identify effects. We acknowledge the existence of these studies and their importance in the general literature. These studies may ultimately be important in assessing the plausibility and magnitude of a dose-response relationship used to quantify effects at various levels of exposure, and/or to identify thresholds. The purpose of this review, however, is to identify positive evidence of impacts, rather than develop dose-response relationships. The negative findings may be consistent with expectations for these effects; it would not be reasonable, for example, to expect all pollutants to exert impacts at every concentration for every endpoint evaluated.

Our understanding of air pollution effects on ecosystems has progressed considerably during the past decades. Nevertheless, many substantial gaps remain. In many cases, the gaps reflect an absence of research; however, even for issues that have been studied, uncertainties often remain. Some of the most profound uncertainties arise from certain unavoidable limitations in study design. More specifically, we note that ecological studies of the impacts of air pollutants tend to fall into three categories: laboratory studies, field studies, and modeling efforts. Each of these study types has strengths and limitations, as discussed below.

*Laboratory studies.* One major strength of laboratory studies is their ability to demonstrate causality. By keeping all conditions constant while varying the parameter of interest–for example, pollutant exposure–researchers can unequivocally link pollutants to effects. However, finding an impact on an endpoint in a laboratory setting does not constitute evidence that similar effects necessarily occur in the environment (Chan *et al.* 2003). Reasons for this include:

- Laboratory studies may use exposure regimes that are not representative of environmental conditions, either in terms of severity or duration of exposure, or in terms of exposure route (Boening 2000, Frederick 2000).
- Laboratory studies may study strains or species of organisms that are unlikely to be representative of their wild counterparts, if any. The studies of the effects of dioxins on chickens are one such example.
- Laboratory studies frequently do not evaluate the impact of a pollutant in the context of other co-occurring contaminants or natural stressors such as predation, drought, and competing organisms/species. These factors may increase or decrease the effect of the pollutant on the endpoint, relative to what is measured in a laboratory.

*Field studies.* Evaluating impacts of pollutants on endpoints through a field study can address many of these concerns; however, field studies are frequently subject to other limitations. For example, field studies are often characterized by a high degree of natural variability, making it difficult to detect biologically significant effects even when present. Furthermore, even when researchers find correlations between exposures and endpoints, confounding factors can make it difficult to establish causality (Chan *et al.* 2003). Due to logistical challenges, field studies also tend to be relatively few in number, and for some endpoints of interest, field study counterparts to laboratory studies are not technically possible.

*Modeling studies.* It is particularly difficult to study endpoints at the larger levels of biological organization (*e.g.,* at the population, community, and ecosystem-level). Geographic areas are larger, and timeframes are longer, rendering it difficult to obtain data in sufficient quantity to detect impacts unless they are exceptionally severe. Therefore, the most common approach to study endpoints at these hierarchical levels is to develop a model. Models may be calibrated using data from laboratory or field experiments and are useful tools in predicting larger-scale, longer-term impacts. However, verifying the predictions and assessing the overall validity of the model can be challenging.

Our review of the impacts of criteria pollutants on ecological impacts reflects work from laboratory, field, and modeling efforts. It is important for readers to recognize, however, that the studies underlying the findings have limitations, and findings should not generally be extrapolated beyond the boundaries of the particular focus of the study. We encourage readers to refer to the original literature for a more complete understanding of the state of the science.

### EFFECTS OF ATMOSPHERIC POLLUTANTS ON NATURAL SYSTEMS

Ecosystem impacts can be organized by the pollutants of concern and by the level of biological organization at which impacts are directly measured. We attempt to address both dimensions of categorization in this overview. In Exhibit 1 we summarize the major pollutants of concern, and the documented acute and long-term ecological impacts associated with them. We follow with a description of each of the major pollutant classes, discussing sources, ecological effects, and identifying sensitive ecosystems.

**EXHIBIT 1 CLASSES OF POLLUTANTS AND ECOLOGICAL EFFECTS**

| POLLUTANT CLASS | MAJOR POLLUTANTS AND PRECURSORS | ACUTE EFFECTS | LONG-TERM EFFECTS |
|---|---|---|---|
| Acidic deposition | Sulfuric acid, nitric acid<br>Precursors: Sulfur dioxide,<br>nitrogen oxides | Direct toxic effects to plant leaves and aquatic organisms. | Progressive deterioration of soil quality due to nutrient leaching, and reduced forest health. Acidification of surface waters. Enhancement of bioavailability of toxic metals (mercury and aluminum) to aquatic biota. |
| Nitrogen Deposition | Nitrogen compounds (*e.g.*, nitrogen oxides, ammonia) | | Nitrogen saturation of terrestrial ecosystems, causing nutrient imbalances and reduced forest health. Soil and water acidification. Progressive nitrogen enrichment of coastal estuaries causing eutrophication. |
| Hazardous Air Pollutants (HAPs) | Mercury, dioxins | Direct toxic effects to animals. | Conservation of mercury and dioxins in biogeochemical cycles and accumulation in the food chain. Sublethal impacts. |
| Ozone | Tropospheric ozone<br>Precursors: Nitrogen oxides and volatile organic compounds (VOCs) | Direct toxic effects to plants. | Alterations of ecosystem wide patterns of energy flow and nutrient cycling; community changes. |

### ACIDIC DEPOSITION

The predominant chemicals associated with acidic precipitation are sulfuric and nitric acid ($H_2SO_4$ and $HNO_3$). These strong mineral acids are formed from sulfur dioxide ($SO_2$) and nitrogen oxides ($NO_x$) in the atmosphere–*i.e.*, the acids are secondary pollutants.

Sulfur compounds are emitted from anthropogenic sources in the form of sulfur dioxide and, to a much lesser extent, primary sulfates, principally from coal and residual-oil combustion and a few industrial processes (NAPAP 1991). Since the late 1960s, electric utilities have been the major contributor to $SO_2$ emissions (NAPAP 1991, EPA 2000).

The principal anthropogenic source of $NO_x$ emissions is fuel combustion (EPA 2003a). Such combustion occurs in internal combustion engines, residential and commercial furnaces, industrial boilers, electric utility boilers, engines, and other miscellaneous sources. Because a large portion of anthropogenic $NO_x$ emissions come from transportation sources (*i.e.*, non-point source pollution), $NO_x$ sources are on average more dispersed compared with anthropogenic sources of $SO_2$ (NAPAP 1991). In the atmosphere, $SO_2$ and $NO_x$ are converted to sulfates and nitrates, transported over long distances, and deposited over large areas downwind of point sources or in the vicinity of urban areas. While emissions of $SO_2$ and $NO_x$ are highest in the Midwestern United States, prevailing winds from west to east cause pollutants emitted in the Midwest to be deposited in New England and Canada (EPA 2000).

Initially, it was thought that $SO_2$ emissions were the only significant contributor to acidic deposition. Subsequently, $SO_2$ emissions decreased, but acidic deposition did not, and the role of nitrogen as a contributor to aquatic acidification became apparent. While initially it was thought that most deposited nitrogen would be taken up by biota, subsequent research indicated that terrestrial ecosystems can become nitrogen saturated, and nitrates then leach out of terrestrial systems into groundwater and streams, causing aquatic acidification (Aber *et al.* 2001, 2003; Driscoll *et al.* 2003a; Fenn *et al.* 1998, 2003; Likens *et al.* 1996, 1998). Moreover, recent findings also suggest that nitrogen is quantitatively as important or, in some areas, more important than sulfur as a cause of episodic acidification in streams (NAPAP 1998, Wigington *et al.* 1996b).

Substantial changes in U.S. sulfur emissions have occurred over the past century, with an increase from 9 million metric tons in 1900 to a peak of 28.8 million tons in 1973, and a subsequent decline to 17.8 million tons in 1998 (Driscoll *et al.* 2001). The decrease in sulfur dioxide emission in recent years was accelerated by Phase I of the 1990 CAAA, implemented in 1995 (EPA 2000, 2003a). The reduction in emissions has been accompanied by both a reduction in atmospheric deposition of sulfate ($SO_4^{2-}$)[1] (EPA 2003a), and a reduction of sulfate in surface waters within the most acid-sensitive regions in the U.S. and Europe (Davies *et al.* 2005; Driscoll *et al.* 2001; EPA 2003b).

Between 1900 and 1990, $NO_x$ emissions in the United States increased from about 2.4 million metric tons to about 21.4 million metric tons, and since then have remained fairly constant (EPA 2003a). Slight decreases in atmospheric nitrogen (N) deposition have been seen in the Northeast over the last 30 years, while a slight increase has been seen in the Upper Midwest (EPA 2003a). Most of these changes were attributed to changes in nitrate ($NO_3^-$) deposition. Importantly, atmospheric deposition of base cations ($Ca^{2+}$, $Mg^{2+}$), which are important in neutralizing acids, showed no significant changes over the last decade, although a slight increase was seen in the Upper Midwest (EPA 2003a). This is important in that the change in acidity has not been accompanied by a change in acid-buffering cations.

Despite recent reductions in U.S. sulfur emissions, in many areas there have not been definitive improvements in pH, acid neutralizing capacity (ANC)[2], or other metrics of surface water acidification (Jeffries *et al.* 2003; Stoddard *et al.*1999, EPA 2003b; Davies

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[1] In water, sulfuric acid ($H_2SO_4$) dissociates into a hydrogen ion ($H^+$) and a sulfate ion ($SO_4^{2-}$).

[2] ANC is a measure of the capacity to buffer or neutralize inputs of strong acid. ANC depends on the availability of base cations ($Ca^{2+}$, $Mg^{2+}$). This in turn depends on the rates of mineral weathering, cation exchange, and immobilization of $SO_4^{2-}$ and nitrogen compounds ($NO_3^-$, $NH_4^+$) in soil.

*et al.* 2005; Likens *et al.* 1996, 1998). Though ANC increased somewhat in some acid-sensitive regions of the U.S. (Adirondacks, Northern Appalachian Plateau and Upper Midwest) between 1990 and 2000 (EPA 2003b), similar changes were not seen in New England or in the Blue Ridge Mountains (EPA 2003b), nor in a number of acid-sensitive streams and lakes in Europe and Canada (Stoddard *et al.* 1999; Jeffries *et al.* 2003; Likens *et al.* 1998). The lack of correlation between $SO_4^{2-}$ reduction and pH or ANC in surface waters reflects the variety of factors that influence surface water chemistry and acid neutralizing capacity.

EPA (2003b) identified five factors involved in determining the rate of recovery of surface waters from acidification:

- Base cations – declining surface water concentrations of base cations ($Ca^{2+}$, $Mg^{2+}$) have occurred in many regions. At some sites, further acidification has occurred despite reductions in sulfate deposition because base cations are declining more rapidly than is sulfate. This loss of base cations limits the magnitude of surface water recovery because of their importance in acid buffering.

- Nitrogen – continued atmospheric deposition of nitrogen may be influencing the acid-base status of watersheds in as-yet undetermined ways. Where watersheds are nitrogen saturated, nitrates can leach into surface waters. Nitrate contributes to acidification of surface waters so its continued presence in atmospheric deposition reduces the rate of recovery of surface waters, despite reduced $SO_4^{2-}$ concentrations.

- Natural organic acidity – increased dissolved organic carbon in acid-sensitive waters may have contributed additional natural organic acidity to surface waters, complicating the response to changes in acidic deposition.

- Climate – climatic changes induce variability in surface water chemistry, making it difficult to detect change in surface waters. Climate or climate-related processes (*e.g.*, the amount of snowcover and number of freezing events) that affect mineral weathering rates may counteract recovery by producing declines in base cations to offset a decline in sulfate, or by inducing an increase in natural organic acidity.

- Lag in response – measuring the response to changes in atmospheric deposition may take longer than the timeframe of available data. Recovery itself may have an inherent lag time, and the changes observed may not be unidirectional.

Because recovery from acidification is a complex process involving these and other factors, the timing and extent of recovery expected under reduced acidic deposition is difficult to predict. At present it is clear that recovery does not closely track changes in acidic deposition, and that there is likely a lag between reduced acidity and recovery of biological communities (Driscoll *et al.* 1998; Jeffries *et al.* 2003; Likens *et al.* 2002).

### ECOLOGICAL EFFECTS

Acidification of ecosystems has been shown to cause direct toxic effects on sensitive organisms as well as long-term changes in ecosystem functions (Exhibit 2). Acidification can affect all levels of biological organization in both terrestrial and aquatic ecosystems. Adverse effects seen in terrestrial ecosystems can the include acutely toxic impacts of

## EXHIBIT 2 EFFECTS OF ACIDIFICATION ON NATURAL SYSTEMS AT VARIOUS LEVELS OF ORGANIZATION

| SPATIAL SCALE | TYPE OF INTERACTION | EXAMPLES OF EFFECTS | | EXAMPLE REFERENCES |
|---|---|---|---|---|
| | | FOREST ECOSYSTEMS | STREAMS AND LAKES | |
| Molecular and cellular | Chemical and biochemical processes | Damages to epidermal layers and cells of plants through deposition of acids. | Decreases in pH and increases in aluminum ions cause pathological changes in structure of gill tissue in fish. | 1, 15, 18 |
| Individual | Direct physiological response | Increased loss of nutrients via foliar leaching. | Hydrogen and aluminum ions in the water column impair regulation of body ions. | 6, 10, 15, 18 |
| | Indirect effects: Death due to ionoregulatory failure. Acidification can indirectly affect response to altered environmental factors or alterations of the individual's ability to cope with other kinds of stress. | Cation depletion in the soil causes nutrient deficiencies in plants. Concentrations of aluminum ions in soils can reach phytotoxic levels. Increased sensitivity to other stress factors like pathogens and frost. | Aluminum ions in the water column can be toxic to many aquatic organisms through impairment of gill regulation. Acidification can indirectly affect submerged plant species, because it reduces the availability of dissolved carbon dioxide ($CO_2$). | 5, 6, 10, 15, 18, 23 |
| Population | Change of population characteristics like productivity or mortality rates. | Decrease of biological productivity of sensitive organisms. Selection for less sensitive individuals. Microevolution of resistance. | Decrease of biological productivity and higher mortality of sensitive organisms. Selection for less sensitive individuals. Microevolution of resistance. | 2, 3, 5, 6, 17, 18, 23, 29 |
| Community | Changes of community structure and competitive patterns | Alteration of competitive patterns. Selective advantage for acid-resistant species. Loss of acid sensitive species and individuals. Decrease in productivity. Decrease of species richness and diversity. | Alteration of competitive patterns. Selective advantage for acid-resistant species. Loss of acid sensitive species and individuals. Decrease in productivity. Decrease of species richness and diversity. | 4, 8, 9, 10,13, 15, 17, 18, 23, 24 |
| Local Ecosystem (*e.g.*, landscape element) | Changes in nutrient cycle, hydrological cycle, and energy flow of lakes, wetlands, forests, grasslands, etc. | Progressive depletion of nutrient cations in the soil. Increase in the concentration of mobile aluminum ions in the soil. | Acidification of lakes and streams. Decrease in acid neutralizing capacity. Persistent acidic conditions in lakes and streams in some regions, despite reduction in sulfate deposition. | 7, 11, 12, 13, 14, 16, 19, 20, 21, 22, 27,28 |
| Regional Ecosystem (*e.g.*, watershed) | Biogeochemical cycles within a watershed. Region-wide alterations of biodiversity. | Leaching of sulfate, nitrate, aluminum, and calcium to streams and lakes. Acidification of aquatic bodies. | Additional acidification of aquatic systems through processes in terrestrial sites within the watershed (nitrogen saturation of terrestrial ecosystems leads to increased nitrate leaching to surface waters). Persistent acidic conditions in lakes and streams in some regions, despite reduction in sulfate deposition. | 8, 11, 12, 13, 14, 16, 22, 25, 26, 28 |


References:

1. Baelstrini *et al.* 2001
2. Baker *et al.* 1996
3. Bobbink and Lamers 2002
4. Boggs *et al.* 2005
5. Bulger *et al.* 2000
6. DeHayes *et al.* 1999
7. Driscoll *et al.* 2001
8. Driscoll *et al.* 2003a
9. Driscoll *et al.* 2003b
10. Elvir *et al.* 2006
11. Hogberg *et al.* 2006
12. Hornbeck *et al.* 1997
13. Innes and Skelly 2002
14. Jeffries *et al.* 2003
15. Laudon *et al.* 2005
16. Lawrence *et al.* 1999
17. Ledger and Hildrew 2005
18. Legge and Kruppa 2002
19. Likens *et al.* 1996
20. Likens *et al.* 1998
21. Likens *et al.* 2002
22. Lovett *et al.* 2000
23. MacAvoy and Bulger 2005
24. McMaster and Schindler 2005
25. NPS 2004
26. Sharpe 2002
27. Stoddard *et al.* 1999
28. EPA 2003b
29. Van Sickle *et al.* 1996

acids on terrestrial plants or, more importantly, chronic acidification of terrestrial ecosystems leading to nutrient deficiencies in soils, aluminum mobilization, and concomitant decreases in health and biological productivity of forests (Driscoll *et al.* 2001, 2003; Likens *et al.* 2001; Mitchell *et al.* 2003). Acidification-induced effects on surface waters are mediated by changes in water chemistry including reductions in ANC and increased availability of aluminum ($Al^{3+}$), which in turn can cause elevated mortality rates of sensitive species, changes in the composition of communities, and changes in nutrient cycling and energy flows. The following paragraphs describe these impacts in more detail.

### Effects on Terrestrial Ecosystems

Acidic deposition increases the concentrations of protons ($H^+$) and strong acid ions ($SO_4^{2-}$, $NO_3^-$) in soils. If the supply of base cations is sufficient to buffer the added acidity, the acidity of soil water will be effectively neutralized. However, if the supply of base cations is low, then atmospheric deposition will cause acidification, which in turn results in the leaching of aluminum ($Al^{3+}$) and nutrients (*e.g.*, sulfate and nitrate) from the soils into surrounding waterways (Driscoll *et al.* 2003c). Leaching of nitrate from soils can contribute to eutrophication of coastal waters, as described in a subsequent section of this report.

Acidification of soils also results in the loss of essential cations from soils, including calcium, magnesium, and potassium ($Ca^{2+}$, $Mg^{2+}$, $K^+$). Soil cation depletion occurs when nutrient cations are displaced from the soil at a rate faster than they can be replenished by slow mineral weathering or deposition of nutrient cations from the atmosphere (Driscoll *et al.* 2001; Likens *et al.* 1996, 1998).

Depletion of cations from soils can lead to a nutrient imbalance in trees and tends to make certain species more susceptible to insect infestation, disease or drought (Driscoll *et al.* 2003c, Nordin *et al.* 2006, Strengbom *et al.* 2006, Throop 2005). Changes in plant physiology and metabolism (Legge and Krupa 2002), resulting in changes in allocation of biomass and nutrients within plants (Fenn *et al.* 2003, Burns 2004) also occur. Nutrient imbalance in foliage (Driscoll *et al.* 2003a, 2003b; Elvir *et al.* 2006; Mitchell *et al.* 2003; DeHayes *et al.* 1999) and changes in epicuticular wax structure (Balestrini and Tagliaferri 2001) have also been documented. All of these can lead to changes in individual plant survival, as well as changes in forest populations and communities.

It is rare for acid deposition to cause direct toxic effects to plants (*i.e.*, phytotoxic effects). Such effects generally only occur at very low pH values, characteristic of areas near smelters and other point sources of sulfur (Legge and Krupa 2002), or in laboratory experiments, where exposures are increased intentionally to examine adverse effects. However, where they occur, toxic effects include injury to leaf epidermal cells and loss of nutrients via foliar leaching (Ashenden 2002). Exposure to high levels of $SO_2$ can also cause reduced photosynthesis, increased in cell wall rigidity, and reduced carbon assimilation (Legge and Krupa 2002).

### Effects on Aquatic Ecosystems

Acidic deposition has resulted in increased acidity in surface waters, especially in areas where acid buffering capacity of soils is reduced and nitrate and sulfate have leached to surface waters. As surface waters acidify, pH levels and ANC both decrease, causing adverse effects to fish and other aquatic biota. While many fish species are acid-sensitive, the main lethal agent is the increase in dissolved aluminum that occurs with falling pH levels (Bulger *et al.* 1998, Van Sickle *et al.* 1996).

Decreased pH and elevated aluminum increase mortality rates of sensitive aquatic species, cause reductions in species diversity and abundance, and cause shifts in community structure (NAPAP 1991; Driscoll *et al.* 1998, 2001, 2003b; Stoddard *et al.* 1999). In some regions of the United States (*i.e.,* New England and acid-sensitive regions of southeastern states), lakes and streams are not chronically acidified but do undergo periodic acidification (Laudon *et al.* 2005; Van Sickle *et al.* 2003; Wigington *et al.* 1996a, b; Van Sickle *et al.* 1996; Vertucci and Corn 1996). This "episodic" acidification involves short-term (hours to weeks) reductions in pH and ANC associated with snowmelt or extreme rainfall events. Acidification episodes have caused increased mortality in fish species including brook trout (*Salvenlinus frontalis*) in Adirondack streams (Van Sickle *et al.* 1996), where the risk of exposure to harmful pH levels during these episodic events is as high as 80 percent for some sensitive fish species (Gerritsen *et al.* 1996).

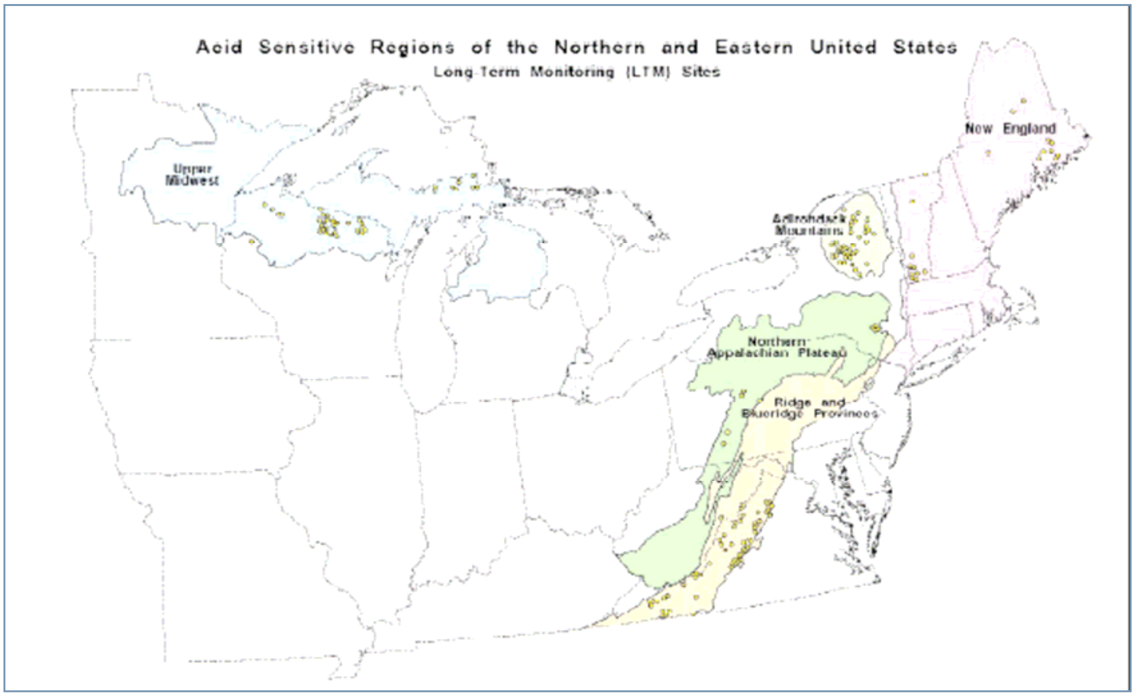
The observed response of both terrestrial and aquatic communities to acidic deposition depends on exposure intensity and duration as well as a host of biotic and abiotic factors. Biotic factors include the genetic make-up, developmental stage, and nutrient status of species, as well as incidence of pathogens and disease. Abiotic factors include soil or water nutrient status, availability of acid-buffering cations, temperature, radiation, precipitation and presence of other pollutants (Legge and Kruppa 2002). These, along with land use history, also influence the general response of ecosystems to acidic deposition (Innes and Skelly 2002). Though the response is variable, adverse effects have been observed and biological recovery is likely to lag behind physical and chemical changes under reduced acid deposition.

### ECOSYSTEMS AT RISK

Ecosystems at risk are those with high acidic deposition and low acid neutralizing capacity. Many of these ecosystems occur downwind of emission sources, often in mountains where soils are thin as well as poorly buffered. High elevation sites are also more vulnerable because mountain fog is frequently more acidic than rain.

Areas in the U.S. with high acid deposition and low acid neutralizing capacity include the southern Blue Ridge Mountains of eastern Tennessee, western North Carolina and northern Georgia; the mid Appalachian Region of eastern West Virginia, western Virginia and central Pennsylvania; New York's Catskill and Adirondack Mountains; the Green Mountains of Vermont; the White Mountains of New Hampshire, and areas of the Upper Midwest (Wisconsin and Michigan) (EPA 2003b, see Exhibit 3).

**EXHIBIT E-3 ACID SENSITIVE REGIONS OF THE NORTHERN AND EASTERN UNITED STATES**




Source: EPA 2003b

Local and regional impacts of acid deposition have been well-documented. For example, a 1990-1994 survey of surface waters showed that 41 percent of lakes in the Adirondack region of New York showed the effects of acidification: 10 percent of lakes were chronically acidic[3] and 31 percent were considered sensitive to episodic acidification.[4] (EPA 2003). Aluminum concentrations in surface waters in New York and New England are often above levels that are toxic to fish or other organisms, and are generally much higher than concentrations observed in surface waters draining watersheds that receive low levels of acidic deposition (Driscoll *et al.* 2003c).

In parts of New England, the Appalachians, and the southeastern United States, the leaching of calcium and magnesium from soils, and the mobilization of aluminum ($Al^{3+}$) have been observed (*e.g.*, Aber *et al.* 2003, Driscoll *et al.* 2001, 2003a, Likens *et al.* 1996, Hogberg *et al.* 2006, Lawrence *et al.* 1999, Pilkington *et al.* 2005, Sullivan *et al.* 2006). Signs of stress connected to acidic deposition in New England forests have been documented in red spruce (Driscoll *et al.* 2003c, DeHayes *et al.* 1999, Elvir *et al.* 2006). In particular, the loss of nutrient cations ($Ca^{2+}$) reduces cold tolerance and can lead to the freezing of foliage at high elevations (Mitchell *et al.* 2003; DeHayes *et al.* 1999; Driscoll *et al.* 2003a, 2003c; Elvir *et al.* 2006). This has led to the dieback of 25-50 percent of the large canopy red spruce in the White Mountains of New Hampshire, the Green Mountains of Vermont and the Adirondacks of New York (Driscoll *et al.* 2003c). The decline in sugar maple in the eastern U.S. has also been attributed to acidic deposition (Sharpe

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[3] Chronic acidity is defined by measured acid neutralizing capacity always equal to zero – that is, the waterbody has no capacity to buffer acids.

[4] Sensitivity to acidification is defined by ANC between 0 and 50 ueq/L.

2002, Horsley 2000, Driscoll *et al.* 2001) although the relative importance of acidic deposition versus other stressors (drought, insects, prior land use) is debated in the literature (see Sharpe 2002, Innes and Skelly 2002).

In acid-sensitive regions of New York, acidification of lakes and streams has caused reduction in species diversity and abundance of plankton, invertebrates, and fish (Driscoll *et al.* 2003b). In western Virginia, declining fish health and reproduction as well as species diversity has accompanied increasing acidity in streams (NPS 2004).

## NITROGEN DEPOSITION

Nitrogen (N) is a naturally occurring element, and is essential to both plant and animal life. Diatomic nitrogen ($N_2$) is an "unreactive" form of nitrogen that constitutes 78 percent of the Earth's atmosphere, and that plants and animals cannot access directly. In order for organisms to draw on this nitrogen to support their growth, the nitrogen must be "fixed" – that is, converted from the unreactive $N_2$ form to a reactive form such as nitrate ($NO_3$) or ammonia ($NH_3$). The availability of reactive nitrogen species limits plant growth in many terrestrial ecosystems (Matson *et al.* 2002) and also tends to be limiting in marine waters. As such, reactive nitrogen species play an important role in controlling the productivity, dynamics, biodiversity, and nutrient cycling of these ecosystems.

Absent human influence, unreactive nitrogen is converted to reactive forms primarily through fixation by certain plants (*e.g.*, legumes). In 1890, anthropogenic activities contributed only about 16 percent to the total amount of reactive nitrogen created. By 1990, however, human activities more than doubled the amount of reactive nitrogen available annually to living organisms (Galloway and Cowling 2002). The primary human activities that result in reactive nitrogen emissions include farming/agriculture and fossil fuel combustion. In the United States, ammonia is produced and released to the environment in large quantities both through the synthesis and application of inorganic fertilizer, and through the growth of nitrogen-fixing crops such as soybeans, alfalfa, peanuts, and others (Howarth *et al.* 2002). Ammonia emissions specifically to the atmosphere come largely via volatilization from animal wastes (*ibid.*). Anthropogenic nitrogen oxide ($NO_x$) emissions to the atmosphere are generally a result of fossil fuel combustion, with electric power generation and automobiles as the largest two sources (EPA 2003).

While emissions have increased since pre-industrial times, some progress has been made in reducing annual emissions in more recent years. U.S. EPA Emissions Trends (http://www.epa.gov/air/airtrends/sixpoll.html)[5] reports that in the United States, $NO_x$ emissions that have decreased 25 percent since 1990. Ammonia emissions estimates are more uncertain, such that it is difficult to determine trends (EPA 2004b).

### ECOLOGICAL EFFECTS

Increased nitrogen availability due to atmospheric deposition can lead to a variety of changes in ecosystem structure and function (Exhibit 4). Because most terrestrial and coastal ecosystems are nitrogen limited, increased supply of nitrogen in terrestrial systems can stimulate uptake by plants and microorganisms, and increase biological productivity. Moderate levels of nitrogen input can have a "fertilizing" effect, similar to the application of nitrogen fertilizer frequently used in timber production or agriculture.

[5] Viewed 20 September, 2006.

**EXHIBIT 4 EFFECTS OF NITROGEN DEPOSITION ON NATURAL SYSTEMS AT VARIOUS LEVELS OF ORGANIZATION**

| SPATIAL SCALE | TYPE OF INTERACTION | EXAMPLES OF EFFECTS | | EXAMPLE REFERENCES |
|---|---|---|---|---|
| | | FOREST ECOSYSTEMS | ESTUARINE ECOSYSTEMS | |
| Molecular and cellular | Chemical and biochemical processes | Increased uptake of nitrogen by plants and microorganisms.<br><br>With chronic exposure, reduced water retention and photosynthesis in some species. | Assimilation of nitrogen by plants, macroalgae, and microorganisms. | 4, 8, 14, 17, 37, 38 |
| Individual | Direct physiological response. | Increases in leaf- size of terrestrial plants. Change in carbon allocation to various plant tissues. | Increase in growth of marine plants. | 4, 13, 25, 26, 27, 29, 37 |
| | Indirect effects: Response to altered environmental factors or alterations of the individual's ability to cope with other kinds of stress. | Decreased resistance to biotic and abiotic stress factors like pathogens, insects, and frost. Disruption of plant-symbiont relationships with mycorrhizal fungi. | Injuries to marine fauna through oxygen depletion of the environment. Loss of physical habitat due to loss of sea-grass beds. Injury and habitat loss through increased shading. Toxic blooms of plankton. | 9, 25, 26, 27, 37 |
| Population | Change of population characteristics like productivity or mortality rates. | Increase in biological productivity and growth rates of some species. Increase in pathogens. | Increase in biological productivity. Increase of growth rates (esp. of algae and macroalgae). | 5, 6, 8, 15, 16, 17, 18, 20, 22, 37 |
| Community | Changes of community structure and competitive patterns | Alteration of competitive patterns. Selective advantage for fast growing species and individuals that efficiently use additional nitrogen. Loss of species adapted to nitrogen-poor or acidic environments. Increase in weedy species or parasites. | Excessive algal growth. Changes in species composition. Decrease in sea-grass beds. Loss of species sensitive to low oxygen conditions. | 5, 8, 18, 22, 24, 27, 29, 33, 34, 35 |
| Local Ecosystem (*e.g.,* landscape element) | Changes in nutrient cycle, hydrological cycle, and energy flow of lakes, wetlands, forests, grasslands, etc. | Changes in the biogeochemical nitrogen cycle. Progressive saturation of microorganisms, soils, and plants with nitrogen. Mobilization of nitrate and aluminum in soils. Loss of calcium and magnesium from soil. Change in organic matter decomposition rate. | Changes in the nitrogen cycle. Depletion of oxygen, increased shading through algal growth. Water quality reduction (reduced water clarity, reduced oxygen levels). | 1, 3, 14, 15, 16, 18, 19, 21, 22, 23, 25, 26, 27, 28, 30, 33, 35 |
| Regional Ecosystem (*e.g.,* watershed) | Biogeochemical cycles within a watershed. Region-wide alterations of biodiversity. | Leaching of nitrate and aluminum from terrestrial sites to streams and lakes. Acidification of soils and aquatic bodies. Increased emission of greenhouse gasses from soils to atmosphere. | Additional input of nitrogen from nitrogen-saturated terrestrial sites within the watershed. Regional decline in water quality in waterbodies draining large watersheds (e.g. Chesapeake Bay). | 7, 10, 11, 12, 10, 11, 12, 15, 16, 18, 21, 22, 25, 26, 27, 33, 35 |


References:

1. Aber *et al.* 1998
2. Aber *et al.* 2001
3. Aber *et al.* 2003
4. Aldous 2002
5. Bobbink and Lamers 2002
6. Boggs *et al.* 2005
7. Bradford *et al.* 2001
8. Burns 2004
9. Carfrae *et al.*
10. Driscoll *et al.* 2003a
11. Driscoll *et al.* 2003b
12. Driscoll *et al.* 2003c
13. Elvir *et al.* 2006
14. Evans *et al.* 2006
15. Fenn *et al.* 1998
16. Fenn *et al.* 2003
17. Howarth *et al.* 2002
18. Jaworski *et al.* 1997
19. Kang and Lee 2005
20. Magill *et al.* 2000
21. Murdoch *et al.* 1998
22. NOAA 2003
23. Neff *et al.* 2002
24. Nordin *et al.* 2006
25. Paerl 2002
26. Paerl *et al.* 2002
27. Paerl *et al.* 2006
28. Pilkington *et al.* 2005
29. Schwinning *et al.* 2005
30. Sinsabaugh *et al.* 2004
31. Small and McCarthy 2005
32. Saiya-Cork *et al.* 2002
33. Spokes *et al.* 2006
34. Stevens *et al.* 2004
35. Swackhamer *et al.* 2004
36. Throop 2005
37. Valiela *et al.* 1997
38. Van der Heijden *et al.* 2004

In the long run, however, chronic deposition of nitrogen adversely affects organisms, communities, and biogeochemical cycles of watersheds and coastal estuaries.

Nitrogen excess in watersheds can lead to disruptions in plant/soil nutrient relations, increased acidity and aluminum mobility in soil, increased emissions of nitrogenous greenhouse gasses from soil, reduced methane consumption in soil, leaching of nitrate ($NO_3^-$) from terrestrial systems to ground and surface waters, decreased water quality, and eutrophication of coastal marine waters (Fenn *et al.* 1998).

Effects on terrestrial and aquatic communities due to acidification (in part due to atmospheric nitrogen deposition) were discussed previously. The following sections describe over-fertilization and eutrophication effects due to atmospheric nitrogen deposition.

### Effects on Terrestrial Systems

The nitrogen over-enrichment process in terrestrial ecosystems has been described as "nitrogen saturation" (Aber *et al.* 1989, 1998). Nitrogen saturation occurs when the assimilative capacity of plants and soils is reached. The process has been described as occurring in four stages (Aber *et al.* 1989):

- Stage 0: Typical condition of nitrogen limitation.
- Stage 1: Nitrogen concentrations in foliage and possibly tree production increase, with brief periods of excess nitrogen runoff from soils to groundwater and surface waters as the capacity for nitrogen assimilation (uptake by plants and storage in soils) is reached.
- Stage 2: Nitrogen losses (nitrate leaching) from forests sustained; nitrification rate[6] increases; nutrient imbalances in foliage occur due to leaching of soil cations.
- Stage 3: Forests decline, and productivity decreases.

Symptoms of nitrogen saturation have been seen in a number of forests receiving chronic low levels of nitrogen addition (Aber *et al.* 1989, 1998, 2003; Driscoll *et al.* 2003a; Fenn *et al.* 1998, 2003; Likens *et al.* 1996; Hogberg *et al.* 2006; Lawrence *et al.* 1999; Pilkington *et al.* 2005; Sullivan *et al.* 2006).

A key symptom of nitrogen saturation is leaching of nitrate from soils to groundwater and streams as the assimilative capacity of soils and plants is exceeded (Fenn *et al.* 1998; Aber *et al.* 1989, 1998). Additional symptoms of nitrogen saturation in watersheds include higher nitrogen-to-nutrient ratios in foliage (*e.g.*, N:Mg, and N:P ratios), foliar accumulation of amino acids or $NO_3^-$, leaching of nutrients from vegetation, and low carbon-to-nitrogen ratios in soil (Aber *et al.* 2001, 2003; DeHayes *et al.* 1999; Fenn *et al.* 1998). Reductions in productivity and greater mortality of trees may also result from nitrogen over-enrichment (Fenn *et al.* 1998, Innes and Skelly 2002).

Biological community composition can also change under increased nitrogen loads, as species more tolerant of high-nitrogen conditions out-compete those less tolerant.

[6] Nitrification is the process whereby ammonium compounds in dead organic material are oxidized into nitrates and nitrites by soil bacteria, which makes nitrogen available to plants; if plant uptake is saturated and nitrification increases, then nitrates leaching is further enhanced.

Changes in forest (Driscoll *et al.* 2003, Fenn *et al.* 2003, Magill *et al.* 2000, Small and McCarthy 2005) and grassland (Schwinning *et al.* 2005, Stevens *et al.* 2004) communities have been documented. For example, chronic increases in nitrogen availability has led to growth inhibition in pine stands in Massachusetts (Magill *et al.* 2000), decline in red spruce throughout New England (Driscoll *et al.* 2003c), and invasion by weedy species in Colorado grassland communities (Schwinning *et al.* 2005).

Because nitrogen is an important nutrient in biological systems, biogeochemical cycles may change when the nutrient balance is disrupted by excess nitrogen. Such changes include increases in the fluxes of the greenhouse gasses nitric oxide (NO), nitrous oxide ($N_2O$), and methane ($CH_4$), from soils to the atmosphere (Fenn *et al.* 1998, Matson *et al.* 2002). Nitric oxide also contributes to the formation of tropospheric ozone (Matson *et al.* 2002). Both increased emissions of these gasses and reduced storage of $CH_4$ have been correlated with higher nitrogen levels in soil (Bradford *et al.* 2001; Fenn *et al.* 1998).

Other biogeochemical responses to increased nitrogen availability include reduced extracellular enzyme function near plant roots (Kang and Lee 2005) and reduced decomposition of soil organic matter (Sinsabaugh 2004, Saiya-Cork 2002). The reduction in decomposition rates can lead to changes in nutrient turnover and soil formation, both important ecosystem processes.

### Effects on Fresh Waters

Because fresh waters are generally not nitrogen limited, the addition of nitrogen does not lead to excessive eutrophication as it does in coastal waters. However nitrate leaching from terrestrial systems to fresh waters leads to acidification effects, as discussed previously.

### Effects on Coastal Waters

Coastal waters are an extraordinarily important natural resource, providing spawning grounds/nurseries for fish and shellfish, foraging and breeding habitat for birds, and generally contributing greatly to the productivity of the marine environment. Critical to the health of coastal waters is an appropriate balance of nutrients. However, many of our nation's estuaries suffer from an excess of nutrient input, particularly an excess of nitrogen.

If present in mild or moderate quantities, nitrogen enrichment of coastal waters can cause moderate increases in productivity, leading to neutral or positive changes in the ecosystem. However, because coastal waters are generally nitrogen limited, too much nitrogen leads to excess plant and/or algal production, decreasing water clarity and reducing concentrations of dissolved oxygen, a situation referred to as eutrophication (Bricker *et al.* 1999; Howarth *et al.* 2002; Jaworski *et al.* 1997; Howarth *et al.* 2003; Paerl 2002a,b; Pearl *et al.* 2006; Valiela *et al.* 1997). Eutrophication may be accompanied by massive blooms of nuisance and toxic algae, habitat loss for fish and shellfish, alteration of food webs, degradation and loss of seagrass beds, alteration of food webs, and the loss of biological diversity (NRC 2000, Howarth and Paerl 2002a, Valiela *et al.* 1997).

Nitrogen loading has recently been cited as the biggest threat to coastal waters because of its role in eutrophication (Howarth *et al.* 2002, 2003). Nationwide, eutrophication is causing widespread loss of habitat, declining water quality, and changes in species composition.[7] A recent survey of coastal waters showed that symptoms of eutrophication are present in 60 percent of the nation's estuaries, with high expression of symptoms in approximately one third of surveyed coastal waters (Bricker *et al.* 1999; also see Exhibit 5).

**EXHIBIT 5** **U.S. ESTUARIES IMPACTED BY EUTROPHICATION**


Source: NEEA 2007

Nitrogen inputs to coastal waters come from several sources and at some locations may be derived primarily from fertilizer runoff and/or sewage waste streams. However, atmospherically derived nitrogen contributes a sizable proportion of the total nitrogen load to estuaries (Bowen and Valiela 2001; Paerl 1997, 2002a, b; Pearl *et al.* 2006; Howarth *et al.* 2003; Valiela *et al.* 1997). Estimates vary widely (from about 10 to 60 percent), but recent reviews of literature suggest about 20 to 40 percent of total nitrogen load to coastal waters is derived from atmospheric deposition (NRC 2000; Paerl 2002a, 2002b). It is important to note that the airsheds delivering atmospheric nitrogen to coastal waters can be 10 to >30 times greater in size than the corresponding watersheds (Paerl 2002a, 2002b). Therefore, relatively rural coastal areas with no major sources in the watershed can be affected by nitrogen originating in distant metropolitan areas.

Atmospheric nitrogen can enter wetlands and coastal estuaries either directly through deposition to water surfaces or indirectly through deposition to the terrestrial portions of

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[7] Changes in species composition are an effect noted in a number of studies of the impacts of pollution on ecosystems. In many cases, the observed "changes" are in fact reductions in species diversity as the more sensitive species lose the ability to compete, and increases in dominance by one or a few tolerant species. Although authors seldom explicitly comment on the desirability or lack of desirability of "changes" in species composition, when the observed changes represent diversity reductions and/or are alterations relative to a less anthropogenically-influenced state, they are usually taken to be adverse rather than desirable effects.

the watershed (Alexander *et al.* 2000; Bowen and Valiela 2001; Howarth *et al.* 2002; Paerl, 2002a, 2002b; Spokes *et al.* 2006; Swackhamer *et al.* 2004; Valiela *et al.* 1997).

### ECOSYSTEMS AT RISK

Atmospheric N deposition is highest in the northeastern and eastern central regions of the U.S. (Fenn *et al.* 1998, NADP 2000, Driscoll *et al.* 2001). Across most of the western and southern United States substantial elevated nitrogen deposition occurs only in isolated areas or "hot spots" in proximity to large sources. Hot spots occur throughout the U.S., in areas close to intensive livestock production, high-elevation areas on which cloud droplet deposition may contribute substantial N inputs, and urban areas with large $NO_x$ concentrations. At such sites, the local N input from the atmosphere may exceed 50 kg N $ha^{-1}$ (Fowler *et al.* 1999).

Nitrogen deposition patterns, availability of soil cations for buffering acidic forms of nitrogen, biotic community composition, successional stage, and presence of other stressors (*i.e.*, extreme weather, insects, drought) influence the response to nitrogen deposition. High-elevation areas where $NO_x$-rich clouds and snow deposit more nitrogen are more susceptible than other areas (Fenn *et al.* 2003, Lovett and Kinsman 1990).

Fenn *et al.* (1998) described characteristics of terrestrial systems susceptible to nitrogen saturation. The most susceptible ecosystems were found to be mature forests with high soil nitrogen stores and low soil carbon to nitrogen ratios. Additional characteristics favoring low N retention capacity include a short growing season (reduced plant N demand) and reduced contact time between drainage water and soil (*i.e.*, porous coarse-textured soils, exposed bedrock or talus). Specific areas of concern include the high-elevation, non-aggrading spruce-fir ecosystems in the Appalachian Mountains, eastern hardwood forests in West Virginia, and southern California mixed conifer forests and chaparral watersheds with high smog exposure (Fenn 1998, 2003).

In estuaries, the water residence time or flushing rate, along with nitrogen inputs influence the susceptibility to nitrogen-induced eutrophication. Salinity, temperature, and the biotic community characteristics also determine the response to nutrient enrichment. Enclosed embayments, where dilution and flushing to marine waters is reduced, are more susceptible to nutrient loading and eutrophication. Coastal waters along the Mid- and North-Atlantic and Gulf of Mexico coasts tend to have a larger percentage of total nitrogen coming from atmospheric deposition (Howarth *et al.* 2003; Paerl 2002a, 2002b). Therefore enclosed embayments along the Mid-Atlantic and Gulf of Mexico are the areas most at risk of eutrophication due to atmospheric nitrogen deposition.

## HAZARDOUS AIR POLLUTANT DEPOSITION

Hazardous air pollutants (HAPs) are a general category of toxic substances covered under Title III of the Clean Air Act, which lists 189 HAPs. Of these 189 substances, the best understood in terms of the potential for adverse ecological impacts include mercury, polychlorinated biphenyls (PCBs), dioxins, and dichlorodiphenyl-trichloroethane (DDT). The use of PCBs and DDT was effectively illegal in the United States prior to 1990 (EPA 1992), and there are currently no plans for additional CAAA regulations of these compounds (Federal Register Unified Agenda 1998). Regulatory actions have reduced but have not eliminated anthropogenic emissions of the remaining two toxins, mercury and dioxins.

## MERCURY

Mercury (Hg) is a toxic element found ubiquitously throughout the environment. The sources of mercury to the biosphere can be grouped as follows (UNEP 2002):

- Natural sources, such as volcanic activity, forest fires, and weathering of rocks;
- Current/ongoing anthropogenic activities, such as fossil fuel combustion, leaks from industrial activities, and the disposal or incineration of wastes; and
- Re-mobilization of past anthropogenic releases from environmental media such as soils, sediments, waterbodies, landfills, and waste piles.

Many researchers have attempted to quantify both total global atmospheric mercury emissions and to estimate the proportion attributable to anthropogenic sources. Recent estimates of the anthropogenic proportion of atmospheric emissions, including both new emissions and re-emissions, generally fall between 50 and 80 percent (EPA 1997, Seigneur *et al.* 2004). There is uncertainty in how much of anthropogenic emissions is attributable to new releases as distinct from remobilization; however, several researchers have estimated these to be approximately equal or at least within a factor of two of each other (Seigneur *et al.* 2004).

Over time, anthropogenic emissions have resulted in increases in the global atmospheric reservoir of mercury. Estimates of the extent of these increases since preindustrial times range from a factor of two to five (EPA 1997, Boening 2000). Once released to the atmosphere, mercury can be transported around the globe, and through wet and dry depositional processes, may contaminate areas far from its point of release. Estimates of the increase in atmospheric deposition of mercury since preindustrial times range from 1.5 to 4, excluding industrial areas where deposition rates are higher (Swain *et al.* 1992, UNEP 2002). North American anthropogenic sources on average contribute roughly 20 to 30 percent of total mercury deposition within the continental United States (Seigneur *et al.* 2004, Selin *et al.* undated). The remainder comes from anthropogenic emissions of other countries and natural sources.

As people have become increasingly aware of the hazards posed by mercury releases, some countries have taken steps to limit emissions (UNEP 2002). In the United States, between the 1990 passage of the CAAA and 1999, anthropogenic emissions have declined by 45 percent. The Clean Air Mercury Rule, issued March 15, 2005, aims at further reducing mercury emissions from coal-fired electric power plants, which are the largest remaining source of mercury emissions in the country.[8]

Many researchers have modeled the fate, transportation, and transformation of mercury both globally and nationally. Within the U.S., the deposition of mercury is highest in areas east of the Mississippi River, particularly in the northeast (EPA 1997, Seigneur *et al.* 2004). Consistent with this result, sampling of common loons indicates a west to east trend in blood and egg methylmercury concentration, with the highest levels occurring in New England and New York (Evers 2004). In general, mercury contamination in aquatic ecosystems of the northeastern United States has been extensively documented (Vanarsdale *et al.* 2005).

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[8] See: http://www.epa.gov/mercury/control_emissions/emissions.htm, viewed 16 August 2006.

Mercury is primarily released in its elemental and inorganic forms. However, it can undergo various transformations in the environment, and its chemical form determines not only its environmental fate but also its potency as a toxicant. From a biological perspective, the most hazardous form of mercury is methylmercury both because of its bioaccumulation and biomagnification potential, but also because organic forms of mercury (including methylmercury) are the most toxic (Wolfe *et al.* 1998, Boening 2000). The main mechanism through which mercury becomes methylated is thought to be through the action of sulfate-reducing bacteria, particularly in freshwater sediments and wetlands (Wiener *et al.* 2003, Evers *et al.* 2005). Many factors affect the rate of mercury methylation in waterbodies, including pH, acid neutralizing capacity, sulfate content, dissolved organic matter, waterbody morphometry, and temperature (Wiener *et al.* 2003, EPRI 2004, EPA 2005b). In contrast, concentrations of methylmercury in soils are generally low (EPA 2005b), and much less is known about mercury uptake and bioaccumulation in terrestrial ecosystems (Rimmer *et al.* 2005).

Methylmercury is the only form of mercury that biomagnifies through food chains (Chan *et al.* 2003, EPRI 2004), with higher trophic level organisms acquiring increasingly large body burdens (EPA 1997, EPA 2005b). Nearly all of the mercury in fish is in the form of methylmercury (Wiener and Spry 1996, EPA 1997, Eisler 2000a).

As of 2004, 21 states had statewide mercury advisories in place for freshwater lakes and/or rivers, and 13 states had statewide mercury advisories for their coastal waters (EPA 2005c). Tribes had two statewide advisories in place, and mercury-based fishing advisories accounted for 76 percent of all advisories (*ibid.*). While acknowledging that local discharges are the impetus behind some of these advisories, these observations nevertheless suggest that atmospheric mercury deposition continues to contribute significantly to mercury levels in aquatic ecosystems nationally.

As an element, mercury does not break down although it does change chemical form. Mercury is likely to persist at levels of concern in ecosystems for some time. Although some atmospherically-released mercury is deposited to terrestrial environments where it is largely sequestered, as mercury accumulates in soils, some amount (less than 30 percent of that which is deposited within a watershed) will be slowly re-released to freshwater bodies and oceans. Modeling efforts by Swain *et al.* (1992, reviewed in Mason *et al.* 1994) suggest that the retention of mercury by some lakes is essentially complete. Studies by Mason *et al.* (1994) predict that elimination of anthropogenic mercury presently in the oceans and in the atmosphere would take 15 to 20 years after the complete termination of all anthropogenic emissions. Because of mercury's persistence in terrestrial and aquatic environments, it will probably take some time for reductions in mercury emissions to be notable in ecosystems (Swain *et al.* 1992, reviewed in Mason *et al.* 1994).

### Ecological Effects

Methylmercury is a potent neurotoxin that at sufficient levels can cause neurologic damage and death in both animals and humans. As indicated in Exhibit 6, adverse effects on wildlife include neurotoxicity, reproductive, behavioral, and developmental effects (EPA 1997). These types of effects have been observed in laboratory studies of mammals, birds, fish, and aquatic invertebrates. While species sensitivity varies, within a species the early life stages are generally the most sensitive (Wiener and Spry 1996, Eisler 2000a, Boening 2000).

**EXHIBIT 6 EFFECTS OF MERCURY ON NATURAL SYSTEMS AT VARIOUS LEVELS OF ORGANIZATION**

| SPATIAL SCALE | TYPE OF INTERACTION | EXAMPLES OF EFFECTS | EXAMPLE REFERENCES |
|---|---|---|---|
| Molecular and cellular | Chemical and biochemical processes | Mercury enters the body of vertebrates and binds to sulfhydryl groups (*i.e.*, proteins). Altered activities of certain enzymes in the blood, liver, and brain. Altered density of certain receptors in the brain. | 1, 3, 4, 8, 9, 12 |
| Individual | Direct physiological response. | Neurological effects in vertebrates. Depending on species, impacts may include: loss of appetite, tissue histopathology including brain lesions, prey capture impairment, reduced foraging, inability to feed, weight loss, metamorphosis inhibition, lethargy, muscular incoordination, and altered incubation behavior. | 3, 4, 5, 6, 7, 11, 12 |
| | Indirect effects: Response to altered environmental factors or alterations of the individual's ability to cope with other kinds of stress. | Few effects known.<br>Damages through increased sensitivity to other environmental stress factors could occur, for example, through impairment of immune response. | 12 |
| Population | Change of population characteristics like productivity or mortality rates. | Reduced reproductive success of fish and bird species. Increased mortality rates, especially in earlier life stages. | 2, 4, 5, 6, 7, 10, 12 |
| Community | Changes of community structure and competitive patterns | Not well understood. | |
| Local Ecosystem (*e.g.*, landscape element) | Changes in nutrient cycle, hydrological cycle, and energy flow of lakes, wetlands, forests, grasslands, etc. | Not well understood. | |
| Regional Ecosystem (*e.g.*, watershed) | Biogeochemical cycles within a watershed. Region-wide alterations of biodiversity. | Not well understood. | |

References:
1. Basu *et al.* 2005
2. Boening 2000
3. Chan *et al.* 2003
4. Eisler 2000
5. Evers 2004
6. Evers *et al.* 2004
7. Frederick 2000
8. Hoffman and Heinz 1998
9. Hoffman *et al.* 1998
10. Meyer *et al.* 1998
11. Wiener and Spry 1996
12. Wolfe *et al.* 1998

Most studies of mercury's effects have been laboratory dosing studies; field studies are relatively few. Furthermore, most studies have focused on aquatic or aquatically-linked organisms, such as fish species, mink, and loons, presumably because of the higher rates of methylation in aquatic ecosystems and consequent potential for higher bioavailability of methylmercury to these organisms. Considerably less research has been devoted to effects on terrestrial species or plants, although some studies have found evidence of impacts to photosynthesis and transportation, water uptake, chlorophyll synthesis, and root damage (Boening 2000).

Impacts have been observed at several levels of biological organization. At the molecular level, mercury interacts with reduced sulfhydryl groups (Chan *et al.* 2003). Sulfhydryl groups are part of many proteins and enzymes; thus, methylmercury may interfere with the actions of these structures, directly or indirectly altering cellular metabolism. The literature has documented impacts of methylmercury on the activity of certain enzymes, including several enzymes present in the brain (Hoffman and Heinz 1998, Wolfe *et al.* 1998).

Laboratory dosing studies of fish and shellfish have found that mercury reduces growth, increases tissue histopathology, and impairs olfactory receptor function (Eisler 2000a). Neurotoxicity-related symptoms in fish include incoordination, inability to capture prey, diminished responsiveness, emaciation, brain lesions, and death (Wiener and Spry 1996, Eisler 2000a). Species investigated in these studies have included rainbow trout, brook trout, catfish, amphipods, mummichog, fathead minnows, mysid shrimp, and others (Eisler 2000a). The extent to which these effects occur under in the wild is unclear, although at most locations acutely toxic effects are unlikely as ambient mercury concentrations are generally lower than those found to cause effects in dosing studies (EPA 2005b).

Dosing studies of bird species have also found evidence of neurotoxicology, ranging from blood and tissue chemistry changes to brain lesions, reduced growth, developmental alterations, behavioral alterations, reproductive impairment, and death (Frederic 2000, Eisler 2000a). Reproductive effects include not only embryomortality and development but also appear to extend to juvenile survival (Wolfe *et al.* 1998). Avian species investigated include mallards, quail, ring-necked pheasants, chickens, house sparrows, northern bobwhite, goshawks, red-tailed hawks, and others (Thompson 1996, Eisler 2000a). Overall, dietary mercury concentrations of about 10 ppm wet weight have been lethal to various species, and egg concentrations of 2 ppm are associated with detrimental effects (Thompson 1996), although species vary in sensitivity.

Although a number of correlative studies of mercury and various avian reproductive endpoints under field conditions did not find effects or were subject to confounding factors (Thompson 1996), there are several examples that strongly suggest mercury is adversely impacting at least some species in some locations. The common loon is probably the best studied in this regard. Effects associated with field exposure to mercury in this species include elevated corticosterone hormone levels, reduced foraging behavior, reduced incubation activity, and reduced fledgling production (Evers *et al.* 2004).

Methylmercury causes neurotoxic effects in mammals, including brain lesions, ataxia, anorexia, disorientation, paralysis, and death (Wolfe *et al.* 1998, Frederick 2000). Dansereau *et al.* (1999) found a tentative link between methylmercury in the diet of mink and whelping rates. As for other species groups, most research has been in the form of laboratory studies, and the most commonly evaluated non-domestic species are mink and otter (Wolfe *et al.* 1998, Eisler 2000a). There have been only a few scattered incidents of potential mercury toxicity to mammals, most notably the death of a Florida panther (Roelka *et al.* 1991, as cited in Thompson 1996). This same research also suggested a potential impact of mercury exposure on Florida panther kitten survival (*ibid.*). In general, dietary methylmercury concentrations of 1 to 6 ppm wet weight have been shown to be sufficient to cause mercury intoxication in mammals (Thompson 1996, Dansereau *et al.* 1999).

### Ecosystems at Risk

Patterns of atmospheric mercury deposition, the availability of mechanisms and environmental conditions that favor methylation, and the structure of the food web all influence the potential for wildlife exposure to methylmercury. Methylation is thought to be through the action of bacteria present primarily in freshwater sediments, such that these ecosystems are likely to be at higher risk. Factors that render certain aquatic ecosystems to be more mercury-sensitive include low alkalinity or low pH, waters with associated terrestrial areas subject to flooding, dark-water lakes and streams, higher dissolved organic matter concentrations, and anaerobic sediments, amongst other factors (Wiener *et al.* 2003). Wetlands tend to be areas of higher methylmercury production and may contribute methylmercury to associated waterbodies (Wiener *et al.* 2003, EPA 2005b). Watersheds associated with the Great Lakes, the eastern U.S. coast, and the Gulf coast have the highest percentage of wetland land cover (*ibid.*).

Data from the National Lake Fish Tissue Survey and the National Listing of Fish and Wildlife Advisories have generally identified the highest levels of mercury in fish from New England and New York, from southeastern coastal watersheds (North Carolina to Florida), and from scattered other areas around the country (EPA 2005b). Mercury deposition patterns in the United States suggest that northeast ecosystems may be particularly at risk. Because these areas are also subject to acidification, which tends to increase the methylation of mercury, there is concern that freshwater ecosystems in the northeast are likely among the most vulnerable to the effects of mercury (Evers *et al.* 2005). Not surprisingly, a number of the (relatively few) field studies have tended to focus on this region. Certainly many of the studies of mercury impacts on common loons have been in the northeastern United States and in eastern Canadian (*e.g.*, Evers *et al.* 1998, Evers *et al.* 2004, Evers *et al.* 2005).

The Florida Everglades is amongst the better-studied sites with respect to mercury. Although mercury concentrations fall within applicable water quality standards, the local conditions are such that virtually all high trophic level vertebrates have high body burdens (Frederick 2000). There have been no observations of direct toxicity to fish; however, based on measured concentrations and comparisons with laboratory studies, there are grounds to expect that fish may be experiencing behavioral and reproductive impacts, amongst others (Frederick 2000). To date, field studies have not been conducted to explicitly evaluate these endpoints.

Sublethal impacts to birds in the Everglades are likely. For instance, field studies suggest that mercury may predispose juvenile great white herons to disease (Spalding *et al.* 1994 as cited in Frederick 2000). Dosing of great egrets at environmentally realistic levels resulted in impaired immunological responses, reduced appetite, and altered behavior

(Frederick 2000). Altogether, "it is strongly suspected that exposure of nestlings to Everglades diets is likely to result in increased juvenile mortality" (*ibid.*). Population-level impacts are possible, as modeling suggests populations of great egrets are sensitive to changes in juvenile survival (*ibid.*).

Despite this body of work, significant gaps remain in our understanding of mercury in the Everglades. In addition to those noted above, little or no research has investigated potential impacts to plant life or invertebrates, amphibians, or reptiles. The good news is that recent research has suggested that since the mid-1990s when mercury concentrations peaked in the area, mercury levels in this valuable ecosystem declined by about 60 percent, due to declines in local emissions and associated deposition from major local atmospheric sources (Atkeson *et al.* 2005, EPA 2005b).

### DIOXINS

Polychlorinated dibenzo-*p*-dioxins (PCDDs) are a group of 75 organochlorine compounds, often referred to as dioxins (Eisler 2000b). The most toxic member of this group is 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) (*ibid.*). Because TCDD is the most toxic dioxin, the toxicity of a dioxin mixture is often expressed as the *toxic equivalency* (TEQ) of some amount of TCDD (EPA 2005a)[9]. Polychlorinated dibenzofurans (PCDFs) are close chemical relatives of PCDDs. Both classes of compounds are produced by the same processes, and both are ubiquitous in the environment (WHO 1989). TEQ estimates are often given jointly for dioxins and furans. Certain polychlorinated biphenyls (PCBs) exert toxic effects through the same mechanism as dioxins and as such are also frequently included in calculations of TEQs.

Although dioxins can be produced through natural events such as forest fires and volcanic eruptions, most environmental inputs are anthropogenic in origin. EPA (2005a) categorizes dioxin sources into six broad groups: combustion; metals smelting, refining, and process sources; chemical manufacturing; biological and photochemical processes; and reservoir sources (*e.g.*, urban runoff). Between 1987 and 2000, U.S. environmental emissions of dioxins declined by 89 percent, primarily due to reductions in air emissions from municipal waste combustion, medical waste incineration, and cement kilns burning hazardous wastes, as well as wastewater discharged to surface waters from pulp and paper mills (*ibid.*). However, in 2000, of total estimated dioxin releases to the environment, over 90 percent were to the air (*ibid.*).

Dioxins and related compounds are thought to exert most of their toxic effects through interaction with the aryl hydrocarbon receptor (AhR). Dioxins bind to the AhR protein in the cytoplasm of cells. The AhR-dioxin complex then is translocated to the cell's nucleus, where it activates or represses a number of genes (Hahn 2001, Mandal 2005). In laboratory studies, particularly of rodents, TCDD has been shown to cause reproductive toxicity, neurotoxicity, immune suppression, increased inflammatory responses, and cancer (*ibid.*).

Studies in wild species are far fewer, and amongst these, laboratory-based toxicity studies of fish–particularly freshwater fish–dominate the available literature (Boening 1998, Eisler 2000b). Fish are among the most sensitive species to the effects of TCDD, and early life stages are the most vulnerable (Boening 1998, Elonen *et al.* 1998, Tietge *et al.* 1998, Hahn 2001). Young fish exposed to sufficiently high concentrations of TCDD

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[9] This is a draft document, available http://www.epa.gov/ncea/pdfs/dioxin/2k-update/, as of August 23, 2006.

exhibit symptoms resembling blue-sac disease, including edema, hemorrhaging, craniofacial deformity, and death (Elonen *et al.* 1998, Cook *et al.* 2003).

Although fish are amongst the most sensitive species, on a national scale, TCDD levels in wild fish are low relative to the levels thought to be associated with adverse effects to fish (Tietge *et al.* 1998). One past exception to this generality is for fish in the Great Lakes region, particularly Lake Ontario lake trout. Lake trout are amongst the most sensitive fish, and studies of dioxin and dioxin-like contaminant levels[10] in lake trout eggs and sediments predict that TEQs were high enough for some decades to completely eliminate the ability of lake trout fry to survive (Cook *et al.* 2003). This prediction is consistent with field evidence of fry toxicity resembling blue sac disease, and with the decline and eventual extirpation of the lake trout population by 1960 (*ibid.*). However, present exposures are close to predicted "no observable adverse effect levels," and natural reproduction in recent years has improved (*ibid.*).

The risk that dioxins pose to other wildlife is difficult to assess because both laboratory and field studies are few (Boening 1998). The limited available information on the subject includes the following.

In birds, sensitivity to dioxin varies considerably across species, with over 40-fold differences on embryo mortality (Gross *et al.* 2003). Dioxin exposure has been associated with impacts to enzyme activity in species such as the double-crested cormorant, ring-necked pheasant, ring-billed gull, herring gull, and Forster's tern (Eisler 2000b). In wild great blue herons and double-crested cormorants, dioxins have been associated with asymmetric brain development (Henschel 1998). One study (White *et al.* 1994) found that wood duck eggs from a contaminated area had levels of PCDDs and PCDFs 50 times higher than levels in control eggs. The contaminated nests were significantly less successful than control nests, and contaminated ducklings also suffered from teratogenic effects.

During the 1950s and 1950s, fish-eating birds in the Great Lakes region suffered reproductive failures, deformities, and population declines associated with dioxin-equivalents (Grasman *et al.* 1998). The most frequently observed manifestation of this toxicity was characterized by a suite of abnormalities named GLEMEDS, for Great Lakes embryo mortality, edema, and deformity syndrome. However, coplanar PCBs rather than dioxins or furans, contributed the most to the dioxin-like toxicity in this region (Grasman *et al.* 1998).

Impact studies on wild mammalian species are few and have focused primarily on mink. Laboratory dosing studies of adult female mink have found dose-dependant decreases in food consumption and body weights, altered blood chemistry parameters, increased lethargy, and death (Hochstein *et al.* 1998, Hochstein *et al.* 2001). Reproduction was also impacted, with reduced kit birth weights and survival (Hochstein *et al.* 2001). Jaw lesions in kits have also been observed (Render *et al.* 2000).

TCDD and other dioxins are extremely stable chemicals with a persistence that is measured in decades (Boening 1998, Sinkkonen and Paasivirta 2000). For example, Johnson *et al.* (1996) found that, though TCDD levels in fish and sediments from an Arkansas river declined significantly during the 12 years following the initial pollution of

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[10] Although both dioxins and other contaminants contribute to the TEQs in lake Ontario, approximately 60 percent of the predicted TEQs are associated with TCDD in particular (Cook *et al.* 2003).

the river, fish from some locations continued to have levels of TCDD that exceeded Food and Drug Administration (FDA) guidelines. TCDD is subject to photochemical degradation, but since the penetration of light into soils and many natural water bodies is limited, this degradation is slow (WHO 1989, Sinkkonen and Paasivirta 2000). Because of dioxins' toxicity and persistence, their presence is likely to be an issue of concern for decades.

## TROPOSPHERIC OZONE

Ozone is a secondary pollutant formed through the oxidation of volatile organic compounds (VOCs) in the presence of oxides of nitrogen (generically $NO_x$) (Fowler 2002). Tropospheric ozone levels in the northern hemisphere have more than doubled in the last century (Dizengremel 2001), and globally, atmospheric concentrations of tropospheric ozone are increasing at the rate of one to two percent per year (Karkosky 1999, Dizengremel 2001, Barbo *et al.* 2002). In 2000, worldwide average tropospheric ozone levels were approximately 25 percent above thresholds established for damage to sensitive plants (Fiscus *et al.* 2005).

U.S. EPA Trends reports (http:///www.epa.gov/airtrends) state that the United States, VOC and $NO_x$ emissions that contribute to the formation of ground-level ozone have decreased 35 percent and 25 percent, respectively, since 1990. Ozone levels, measured as eight-hour levels, declined by 21 percent in this same time period. However, declines have not been uniform across the United States, and there are a number of counties, particularly in California, where ozone concentrations exceed relevant air quality standards (*ibid.*).

### ECOLOGICAL EFFECTS

Ozone is one of the most powerful oxidants known (Long and Naidu 2002), but its impacts have been little studied in faunal species. The limited available research has found evidence of a variety of pulmonary impacts to specific mammalian and avian species (Rombout *et al.* 1991). In contrast, ozone's impacts on plants are much better understood (*e.g.*, see Exhibit 7). Indeed, EPA (2006a, b, c) provides an extensive review of the impacts of ozone on plants and natural ecosystems, and ozone has been "demonstrated to be the most important phytotoxic pollutant in Europe as well as in North America" (Treshow and Bell 2002).

Researchers have used a variety of controlled exposure studies to evaluate impacts of ozone on plants, usually grown individually but sometimes grown in a group or community setting. Because it is difficult to extrapolate the results of controlled exposure studies to long-term effects on mature forest stands, modeling is often employed to estimate these impacts (Fangmeier *et al.* 2002, Ashmore *et al.* 2002). Field studies also provide valuable information; these efforts tend to evaluate impacts along pollution gradients, examining potential correlations between exposure and endpoints (*e.g.*, Grulke and Balduman 1999, Grulke *et al.* 2001).

**EXHIBIT 7 EFFECTS OF OZONE ON NATURAL SYSTEMS AT VARIOUS LEVELS OF ORGANIZATION**

| SPATIAL SCALE | TYPE OF INTERACTION | EXAMPLES OF EFFECTS | EXAMPLE REFERENCES |
|---|---|---|---|
| Molecular and cellular | Chemical and biochemical processes | Oxidation of enzymes of plants, generation of toxic reactive oxygen species (hydroxyl radicals). Disruption of the membrane potential. Reduced photosynthesis and nitrogen fixation. Increased apoptosis. | 1, 3, 8, 9, 11, 16, 17, 18, 22, 25 |
| Individual | Direct physiological response. | Visible foliar damage, premature needle senescence, altered carbon allocation, and reduced of growth rates. | 1, 2, 3, 4, 8, 11, 13, 14, 15, 17, 20, 22, 24, 25 |
| | Indirect effects: Response to altered environmental factors or alterations of the individual's ability to cope with other kinds of stress. | Increased sensitivity to biotic and abiotic stress factors such as pathogens and frost. Disruption of plant-symbiont relationship (mychorrhizae), and symbionts. | 14, 15, 17, 19 |
| Population | Change of population characteristics like productivity or mortality rates. | Reduced biological productivity and reproductive success. Selection for less sensitive individuals. Potential for microevolution for ozone resistance. | 3, 4, 6, 7, 8, 10, 11, 14, 16, 17, 21, 23, 24 |
| Community | Changes of community structure and competitive patterns | Alteration of competitive patterns. Loss of ozone sensitive species and individuals leading to reduced species richness and evenness. Reduction in productivity. Changes in microbial species composition in soils. | 1, 5, 6, 10, 17 |
| Local Ecosystem (*e.g.,* landscape element) | Changes in nutrient cycle, hydrological cycle, and energy flow of lakes, wetlands, forests, grasslands, etc. | Alteration of ecosystem-wide patterns of energy flow and nutrient cycling (*e.g.,* via alterations in litter quantity, litter nutrient content, and degradation rates; also via changing carbon fluxes to soils and carbon sequestration in soils). | 1, 10, 11, 17 |
| Regional Ecosystem (*e.g.,* watershed) | Biogeochemical cycles within a watershed. Region-wide alterations of biodiversity. | Potential for region-wide phytotoxicological impacts and reductions in net primary production. | 10, 12 |

References:
1. Andersen 2003
2. Andersen and Grulke 2001
3. Ashmore 2005
4. Ashmore 2002
5. Barbo *et al.* 1998
6. Black *et al.* 2000
7. Chappelka 2002
8. Chappelka and Samuelson 1998
9. Dizengremel 2001
10. Felzer *et al.* 2004
11. Fiscus *et al.* 2005
12. Fowler *et al.* 1999
13. Grulke and Balduman 1999
14. Jones *et al.* 2004
15. Karkosky *et al.* 1999
16. Long and Naidu 2002
17. McLaughlin and Percy 1999
18. Miller and McBride 1999
19. Powell *et al.* 2003
20. Rombout *et al.* 1991
21. Takemoto *et al.* 2001
22. Tingey *et al.* 2004
23. Treshow and Bell 2002
24. Vandermeiren *et al.* 2005
25. Weinstein *et al.* 2005

Ozone studies have been conducted on many crops (*e.g.,* beans, corn, cotton, oats, potatoes, rice, soybeans, wheat, alfalfa) and also on a number of tree species, such as ponderosa pine, loblolly pine, Jeffrey pine, quaking aspen, black cherry, red maple, yellow poplar, and northern red oak, amongst others (Ashmore 2002, Barbo *et al.* 2002, Tingey *et al.* 2004, Weinstein *et al.* 2005, also reviewed in EPA 2006b). Ozone sensitivity of plants varies between species, with evergreen species tending to be less sensitive to ozone than deciduous species, and with most individual deciduous trees being less sensitive than most annual plants (EPA 2006b). However, exceptions to this broad ranking scheme certainly exist, and there can be variability not only between species but even between clones of some trees (*ibid.*) and within cultivars (Ashmore 2002). Life stage also matters: in general, mature deciduous trees tend to be more sensitive than seedlings, while the reverse is more typical for evergreen trees (EPA 2006b). The effects of ozone on wild herbaceous or shrub species are less well understood, although available data suggest that some wild species are as susceptible as the most sensitive crops (Ashmore 2002), and it may be reasonable to use crop ozone responses as an analog for the responses of native annual plants (EPA 2006b).

Ozone or its reaction products exert their toxic effects once they reach target plant tissues (EPA 2006b). Ozone reaches plant tissues by diffusing through the stomata of plant leaves (or needles) and interacts with cellular components to generate toxic reactive oxygen species (ROS) such as hydrogen peroxide, superoxide ions, and hydroxyl radicals (Dizengremel 2001), which injure the plant's cells. A classic symptom of ozone exposure is visually-apparent damage to leaves, and such injuries have been observed in both laboratory and field settings (reviewed in Ashmore 2002, Long and Naidu 2002, Ashmore 2005, EPA 2006b). Additional research has shown that ozone stimulates plant defense mechanisms in crops (Fiscus et al 2005).

Ozone exposure impacts other physiological processes in plants, in particular acting to reduce photosynthesis rates (*e.g.,* Fiscus *et al.* 2005, Chappelka and Samuelson 1998, McLaughlin and Percy 1999, Ashmore 2002, Mills 2002) and to increase leaf senescence (Grulke 2003, Long and Naidu 2002, Temple 1999). Reduced photosynthesis and increased leaf loss together reduce plants' ability to generate energy from sunlight. Consistent with these studies, researchers have found that ozone exposure reduces plant growth rates. Reduced growth rates have been observed across a variety of plant species (*e.g.,* Treshaw and Bell 2002, Chappelka and Samuelson 1998, Weinstein *et al.* 2005, Barbo *et al.* 2002; for review see EPA 2006b). For tree species in particular, even small impacts on growth can be important given the cumulative effect over decades (Ashmore 2002).

In addition to reducing overall growth rates, which has potential economic impacts for commercially important species, ozone alters the allocation of resources within the plant. For example, ozone exposure increases carbon (carbohydrate) allocation to leaves and decreases the allocation in roots (Grulke *et al.* 2001, Andersen 2003, Tingey *et al.* 2004; reviewed in EPA 2006b). Reduced allocation of nutrients and biomass to roots may result in indirect effects to impacted plants, including increased susceptibility to root disease, drought, and windthrow (Takemoto *et al.* 2001).

Carbon allocation changes within plants may also reduce the quantity of carbon eventually sequestered in soils (Felzer *et al.* 2004, Fiscus *et al.* 2005). Such changes are important as they are likely to influence the soil-based foodweb, potentially altering carbon retention, mineralization, and other important soil properties (Andersen 2003). Altered nutrient composition in leaves may effect litter quality and decomposition speed, impacting nutrient cycling (*ibid.*).

Ozone exposure also may change plants' allocation of resources between vegetative growth versus seed/flower production (Treshaw and Bell 2002, Black *et al.* 2000, McLaughlin and Percy 1999), potentially impacting long-term reproductive success and population stability. A study of blackberries found ozone exposure to result in the production of smaller, less-ripe fruit (Chappelka 2002). This finding has potential ecological importance, as blackberries help lead to ecosystem recovery in the event of a disturbance and also serve as a major source of food and habitat for many bird and wildlife species (*ibid.*). More broadly, impacts of ozone on reproductive endpoints may result in altered competitive vigor and species composition (Black *et al.* 2000).

Impacts to plant communities may occur as a result of ozone exposure, although such effects have not been studied as extensively due to ecosystem complexity and the long timeframes involved (EPA 2006b). Experiments with an early successional plant community found that ozone reduced vegetative cover, vertical density, species richness, and evenness relative to the control (Barbo *et al.* 1998), although differences were less pronounced in a drought year (*ibid.*). Other observed community level include reducing competitive ability of specific species, changing soil microbial communities, and altering species composition and relative abundance (EPA 2006b).

The effects of exposure to tropospheric ozone may be modified by a variety of environmental factors in the exposed area, including temperature, humidity, light levels, wind speed, and soil nutrient and water content (Mills 2002). Humidity and light levels affect stomatal conductance, resulting in altered within-leaf exposure for a given ambient ozone concentration (*ibid.*). Wind speed also affects the flux of pollutants to the plant by altering the diffusion of the gases between the atmosphere and the leaf surface (*ibid.*). Other factors affecting the plant responses to specific exposures include developmental stage at the time of exposure, plant age, and the presence of other stressors (Andersen and Grulke 2001, Andersen 2003, McLaughlin and Percy 1999, Grulke and Balduman 1999).

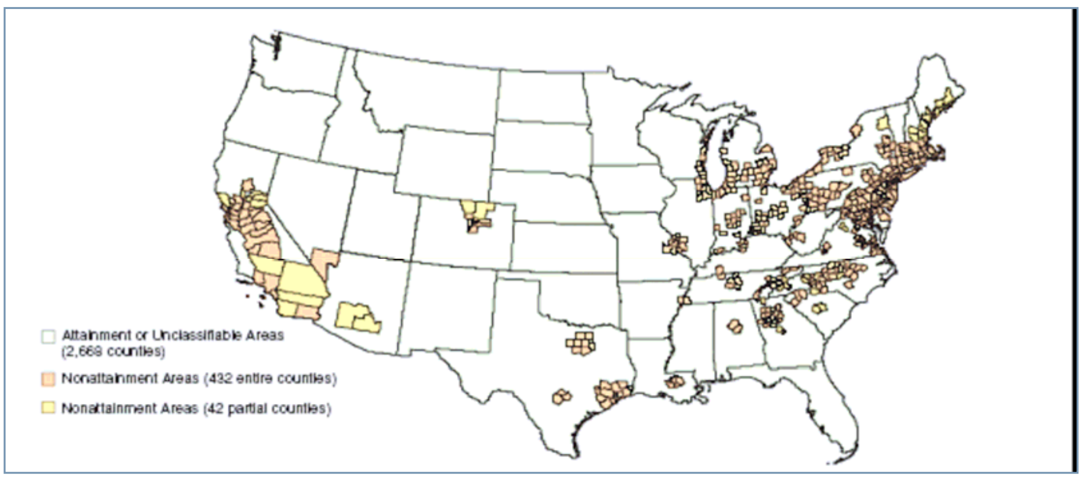
Potential other stressors include additional pollutants. Organisms in ecosystems are seldom exposed to individual pollutants but rather are almost always exposed to a number of compounds, either simultaneously or sequentially (Fangmeier *et al.* 2002). Although relatively more attention has been paid to the interaction between ozone and sulfur dioxide, different experiments have produced different results. It seems that at lower concentrations, these pollutants may interact in a less-than-additive fashion (*i.e.,* antagonistically) with respect to growth and yield, while at higher concentrations, more-than-additive (*i.e.,* synergistic) effects are possible (*ibid.*).

Less research has been conducted on the interactions between ozone and nitrogen, either in the form of gaseous ammonia or nitrogen dioxide. Few clear conclusions are possible with respect to ammonia (*ibid.*). Ozone and nitrogen dioxide applied at environmentally realistic concentrations sometimes did not interact (*i.e.,* effects were additive), and sometimes interactions were antagonistic; at higher concentrations, synergistic impacts to growth and yield appear (*ibid.*). That said, the joint impacts of ozone and nitrogen also may depend on the evaluated endpoint: nitrogen, like ozone, decreases carbon allocation to roots (Grulke *et al.* 1998a as cited in EPA 2006a); however, nitrogen tends to counteract the effect of ozone on photosynthesis (EPA 2006a). Understanding the net effect of such interactions is complex.

### ECOSYSTEMS AT RISK

Researchers have evaluated the distribution of ozone across the United States (Exhibit 8). Counties in the northeast, mid-Atlantic, Midwest, and in California as well as smaller areas in the south and south-central United States are subject to higher levels of ozone (EPA 2004a). The southern Appalachian Mountains also experience particularly high ozone exposures (EPA 2006b). Maximum ozone levels, rather than long-term averages, are of importance because ozone-related damage is thought to be related to maximum exposure values (Treshaw and Bell 2002). Ozone levels vary seasonally and according to the time of day. Concentrations increase in spring through summer months. In industrial areas, ozone levels tend to be higher in the late afternoon, while at marine or high latitude sites, concentrations are highest before sunrise (Felzer 2004).

**EXHIBIT 8** **EPA 8-HOUR OZONE DESIGNATIONS, APRIL 2004**




Source: EPA 2004a

In the field, responses to chronic or recurrent exposure can be subtle and may not observable for many years. In forests, "[o]nly recently have the patterns of change over time (>10 years) at regional and national scales been statistically analyzed with the rigor required to attribute changes in forest health at the species level to individual and/or complexes of biotic and abiotic stressors, including air pollutants" (McLaughlin and Percy 1999).

Ecosystems with known damages attributed to ozone include the San Bernardino Mountains of Southern California and the Sierra Nevada Mountains. Visible oxidant injury to ponderosa pine has been documented in the San Bernardino Mountain range for 40 years (Fenn and Poth 1999). This mixed-conifer forest ecosystem has been exposed to chronically elevated ozone levels over a period of at least 50 years (*ibid.*). The first indications of ozone impacts to the ecosystem were observed as visible damage to

Ponderosa and Jeffrey pines (Miller *et al.* 1963 as cited in EPA 2006b). Direct injuries identified in following years and decades include premature needle senescence, reduced photosynthesis, altered carbon allocation, and reductions in growth rates (Grulke and Balduman 1999; Tingey *et al.* 2004; Miller *et al.* 1982 as cited in McLaughlin and Percy 1999; Temple 1999).

Indirect effects have also been observed in the field, including alterations in trees' interactions with predators, pathogens and symbionts (reviewed in Takemoto *et al.* 2001, EPA 2006b). For example, the accumulation of weakened trees resulted in heavy bark beetle attack that significantly elevated mortality rates during the 1960s and 1970s (reviewed in EPA 2006b, McLaughlin and Percy 1999). Although current ozone levels are lower than previously, higher levels of pollution are still associated with elevated mortality and beetle bark activity (Jones *et al.* 2004).

Altogether, major changes to ecosystem characteristics have occurred, including alterations in species composition, nutrient cycling and energy flows (Arbaugh *et al.* 2003; also reviewed in McLaughlin and Percy 1999, EPA 2006b).

In the Sierra Nevada Mountains, ozone concentrations capable of causing injury have been occurring for many years, but injury has not reached the same proportions as in the San Bernardino forest (Fenn *et al.* 2003, EPA 2006b). The most notable injury in pine stands was chlorotic mottle, which was noted in upwards of 20 percent of trees sampled (Carroll *et al.* 2003). Decreased radial growth in Ponderosa and Jeffrey pine has been observed (reviewed in EPA 2006b). Tree mortality has also been documented, where "one trend plot network in the southern Sierra Nevada recorded seven percent mortality of plot trees due to severe $O_3$ injury over a 23-year period" (*ibid.*). Significant differences in both the forest stand composition (*e.g.,* the presence of fewer conifers and more hardwoods), and site dynamics have probably played an important role in determining the different ecosystem responses (McLaughlin and Percy 1999, Ashmore 2002).

One potentially vulnerable habitat in the event of ozone-influenced reduction of carbon input to soils is the prairie, an ever-shrinking area that is home to exceptionally rich soil (Kline 1997). Whereas forest soils contain approximately 70 tons of organic matter per acre, prairie soils contain as much as 120 tons; indeed, in tallgrass prairies up to 65 percent of the biomass is underground (*ibid.*). Soil carbon in prairies comes largely from plant roots, which decay in place, whereas in forest ecosystems, organic matter enters the soil at the surface, and is transported to a larger extent by the likes of earthworms (*ibid.*). To the extent that ozone alters carbon allocation in prairie plants, prairie soils and soil communities may be particularly impacted.

In these and other areas, ozone may act synergistically with other stress factors to induce further damages to vegetation. For example, in North America, the most frequent occurrence of disease problems is spatially consistent with patterns of the highest levels of ozone and acidic deposition (McLaughlin 1999). Nitrogen deposition is a notable pollutant in both the Sierra Nevada and San Bernardino areas (Fenn *et al.* 2003). Experiments in the San Bernardino area in particular suggest that nitrogen deposition may increase tree susceptibility to beetle attack (Jones *et al.* 2004). In the eastern United States, regionally elevated levels of tropospheric ozone co-occur with high deposition rates of nitrogen, sulfur and acids. These multiple stress factors may have acted synergistically in injuring high elevation forests throughout the eastern United States (Grulke and Balduman 1999, Andersen and Grulke 2001).

## CONCLUSIONS

Predicting ecological impacts of air pollution at the regional scale or for the United States as a whole would require an understanding of effects at many temporal and spatial scales, where there is currently a general lack of data. Furthermore, there is limited transferability of existing information between various spatial and temporal scales and between geographic regions. However, we can reach several general conclusions, based on the existing literature.

- Although ambient concentrations of most air pollutants significantly decreased after the 1990 Clean Air Act Amendments, some pollutants still occur in concentrations high enough to directly injure living organisms. These direct injuries can be observed, for example, in areas with high ambient levels of tropospheric ozone or in some high-elevation ecosystems that are exposed to high levels of acid deposition.

- Air pollutants have indirect effects that can have high ecological importance. Indirect effects include those in which the pollutant alters the physical or chemical environment (*e.g.,* soil properties), the plant's ability to compete for limited resources (*e.g.*, water, light), or the plant's ability to withstand pests or pathogens. Examples are excessive availability of nitrogen, depletion of nutrient cations in the soil by acid deposition, mobilization of toxic elements such as aluminum, and changes in winter hardiness. Indirect effects are more difficult to observe than direct toxic relationships between air pollutants and biota, and there may be a variety of interactions and consequent indirect effects that have not yet been detected.

- There is a group of pollutants that tend to be conserved in the landscape after they have been deposited to ecosystems. These conserved pollutants are transformed through biotic and abiotic processes within ecosystems, and accumulate in biogeochemical cycles. These pollutants include, but are not limited to, hydrogen ions ($H^+$), sulfur- (S) and nitrogen- (N) containing substances, and mercury (Hg). Chronic deposition of these pollutants can result in progressive increases in concentrations and cause injuries due to cumulative effects. Indirect, cumulative damages caused by chronic exposure (*i.e.*, long-term, moderate concentrations) to these pollutants may increase in magnitude over time frames of decades or centuries with very subtle annual increments of change. Examples are nitrogen-saturation of terrestrial ecosystems, cation depletion of terrestrial ecosystems, acidification of streams and lakes, and accumulation of mercury in aquatic food webs.

- Impacts of pollutants are modified by many factors, including species, life stage, environmental chemical and physical factors, and the presence of other pollutants, amongst others. Some information is available about the interactions between these factors, but for the most part, much more research is necessary to understand the complexities involved.

There is evidence that current air pollution is an important environmental stress factor over large areas of the United States and other countries, even if effects have not yet been fully documented. Actions taken now to reduce air emissions may have consequences far into the future and may affect ecosystems both in predictable ways and in ways that are not yet understood.

## SECTION 2: TOOLS AND METHODS TO IMPLEMENT THE ADIRONDACK PARK CASE STUDY OF ECOLOGICAL BENEFITS OF THE CLEAN AIR ACT AMENDMENTS

### I. INTRODUCTION

In July 2003, the EPA Project Team developed a proposed approach to quantifying ecological benefits as part of the anticipated second prospective analysis of the benefits and costs of the CAAA. In May of 2005, the Ecological Effects Subcommittee (EES) of the Advisory Council on Clean Air Compliance responded with an advisory regarding the proposed approach. The EES supported implementing a case study to augment the national level benefits assessments conducted in the first prospective analysis. They suggested a number of upland and coastal sites with service flows potentially affected by CAAA-regulated air pollutants that may be amenable to economic valuation. One such recommendation was to conduct a case study in the Adirondack region of New York State, emphasizing the potential for assessment of impacts to fisheries and timber management.[11]

Adirondack Park is a State Park comprising 5,821,183 acres of State and privately owned land in upstate New York and is nearly a 100 by 100 mile box of land, intersecting fourteen counties.[12,13] The park is defined by the "blue line" which represents the original delineation of the park boundary when it was created in 1892 through an amendment to the State constitution, with the purpose of forest and natural resource conservation.[14] The lands within the "blue line" are often referred to as the Adirondack Ecological Zone. This area is highlighted in Exhibit 1.

This section of the white paper provides an approach for the case study. This section is divided into three major parts; it first outlines the methods and tools that the Project Team will employ to assess the economic benefits of reduced air pollution associated with the Clean Air Act Amendments (CAAA) on recreational fishing in the Adirondacks region of New York State. Second, it describes methods and data for the assessment of economic impacts to the commercial timber industry. Finally, this section summarizes recent research estimating total values for natural resource improvements in the Adirondacks as a result of air policy alternatives, and discusses the intersections of this research with this case study.[15]

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[11] Ecological Effects Subcommittee of the Advisory Council on Clean Air Compliance. Advisory on Plans for Ecological Effects Analysis in the Analytical Plan for EPA's Second Prospective Analysis – Benefits and Costs of the Clean Air Act, 1990-2020.

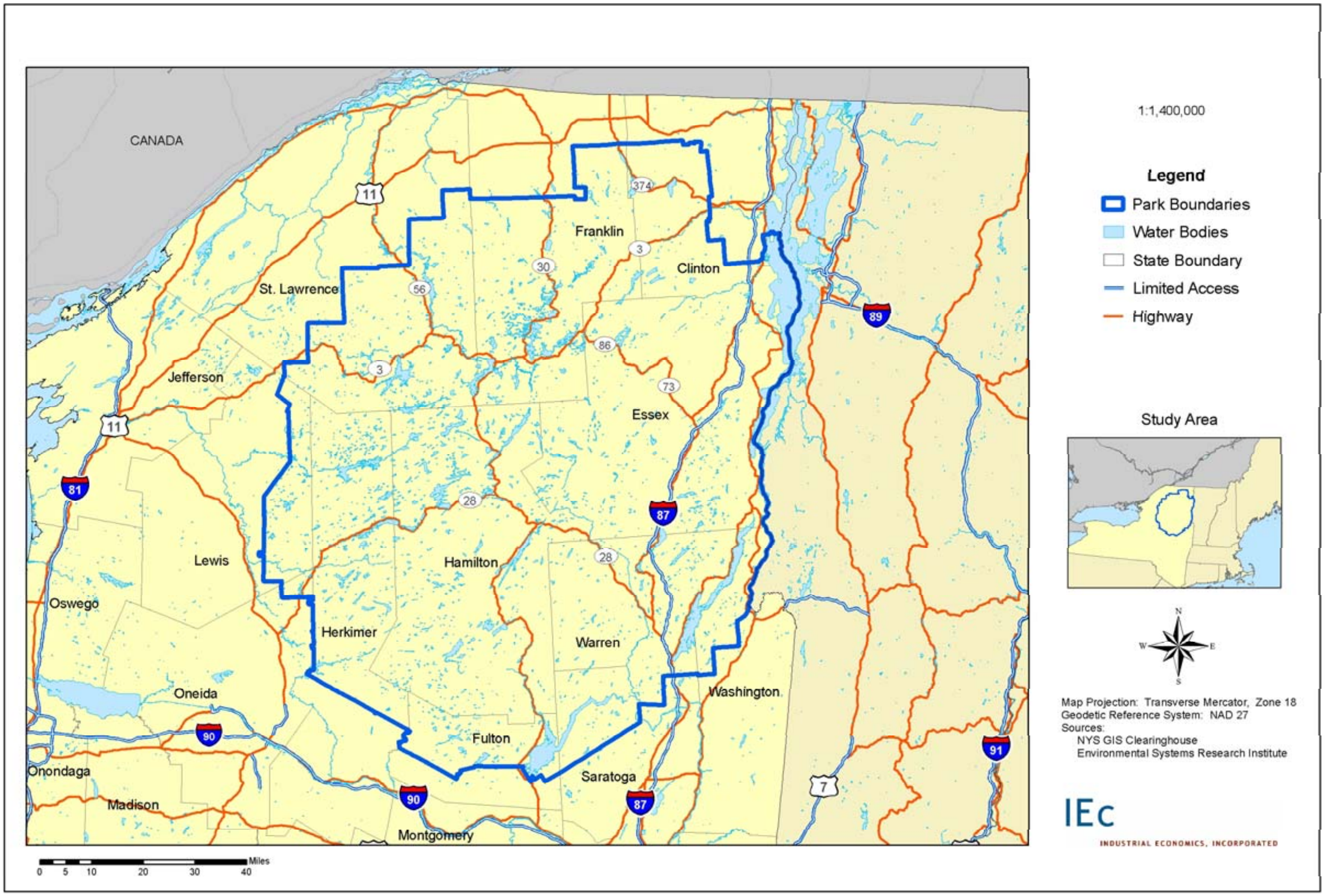
[12] Adirondack Park Agency (APA). 2003. Acreage by County and Land Use Classification. March, 2003. Website accessed August 31, 2006 at: http://www.apa.state.ny.us/gis/colc0303.htm.

[13] Gould D and T Holmes. 2006. The Adirondack Region. Adirondackwood.com website. Accessed on 8/24/06 and available at: http://www.adirondackwood.com/adirondacks.

[14] APA, 2003.

[15] Banzhaf, Spencer et al. September 2004. Valuation of Natural Resource Improvements in the Adirondacks. Resources for the Future.

**EXHIBIT 1. ADIRONDACK PARK BOUNDARIES**



## II. ADIRONDACK CASE STUDY: RECREATIONAL FISHING ANALYSIS

Title IV of the CAAA of 1990 addresses the issue of acidification by mandating reductions in sulfur and nitrogen oxides ($NO_x$ and SOx). This section outlines a case study of the effects of the CAAA on recreational fishing in the Park by measuring the proportion of lakes that remain fishable as a result of the CAAA.

This section first describes the ecological effects of acid rain on surface waters, then characterizes regional recreational fishing activity to provide context for the economic impact analysis. Finally, this section details the methodological tools and approach that will be applied to quantify the impacts of acid rain on recreational fishing.

### ECOLOGICAL EFFECTS OF $NO_X$ AND $SO_X$ DEPOSITION ON FISHERIES

The effects of acidification, defined as deposition of the air pollutants $NO_x$ and $SO_x$, on surface waters, are well-documented. Surface waters, such as lakes and streams, may be the most susceptible systems to acidic deposition as they collect acidic precipitation not only from direct deposition on their surfaces, but also in the form of runoff from their entire watershed. Acid accumulates in surface waters via three main pathways:

- precipitation, or wet deposition, in which pollutants are dissolved in rain or snow;
- dry deposition, or direct deposition of gases and particles on surfaces; and
- cloud-water deposition, involving material dissolved in cloud droplets and deposited on vegetation.[16]

Acidification is described using two measures: pH and acid neutralizing capacity (ANC). The pH is based on the concentration of hydrogen ions in the water; ANC is a measure of the water body's ability to neutralize acids in the water column and is dependent on the specific hydrogeochemical characteristics of the water body, such as water turnover and soil properties. As the ANC approaches zero, the effect of each unit of acid that enters the system increases in terms of reducing the pH. Most freshwater lakes have a natural pH level of between 6.5 and 8.

Acidification results in the gradual loss in the ability of ecosystems to buffer these acids, resulting in changes to ecosystem structure and function. It affects the trophic structure of water contributing to declines in the abundance of zooplankton, macroinvertebrates, and fish.[17] Acidification may impact fish populations directly, or indirectly, through the food chain or reproductive or behavioral impacts. Moderate shifts in pH levels may result in changes in species composition, where acid-sensitive species are replaced by less sensitive species. At more extreme acidification levels, however, species richness, defined as the total number of species occupying a system, may be affected. Research has shown that the number of fish species present is positively correlated with pH and ANC.[18]

### ACIDIFICATION AND RECREATIONAL FISHING IN THE ADIRONDACK PARK

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[16] The U.S. National Acid Precipitation Assessment Program. 1991. Integrated Assessment Report. The NAPAP Office of the Director, Washington, DC.

[17] Driscoll, Charles T. et. al. March 2001. Acidic Deposition in the Northeastern United States: Sources and Inputs, Ecosystem Effects, and Management Strategies. BioScience 51(3): 180-198.

[18] Driscoll, Charles T. et al. 2003. Effects of Acidic Deposition on Forest and Aquatic Ecosystems in New York State. Environmental Pollution 123: 327-336.

The Adirondack Ecological Zone (Park) contains over 3,000 lakes and ponds within the following watersheds that intersect the outer boundaries of the Park: Lake Champlain, Oswegatchie-Black, Raquette, St. Lawrence, Mohawk-Hudson, and Upper Hudson.[19]

The Adirondack region may exhibit the most severe ecological impacts from acidic deposition of any region in North America.[20] Federal programs addressing air pollution have been particularly beneficial to the region as, due to its location downwind of the highly industrialized Ohio River Valley, most of the acid deposition in the region originates from out of state.[21] A 1984 through 1987 survey of Adirondack lakes determined that 27 percent were chronically acidic, and an additional 21 percent were seasonally acidic.[22]

The Adirondack Lake Survey Corporation (ALSC) surveyed 1,469 lakes in the Adirondacks and found that of the 53 fish species recorded, about half are absent from lakes with pH levels below 6. Among the missing species were important recreational fishing species, including Atlantic salmon, tiger trout, redbreast sunfish, bluegill, tiger musky, walleye, alewife, and kokanee.[23] The survey further found that the most abundant were brown bullhead, brook trout, and white sucker, native, acid tolerant species. Most Adirondack fish species prefer neutral or slightly acid conditions. Only four of the 15 most common species were regularly found in lakes with a pH level below 5.5, a marked stress point.[24]

### METHODOLOGICAL APPROACH FOR RECREATIONAL FISHING ANALYSIS

The service flow affected by lake acidification that is most amenable to economic analysis is recreational fishing. Extensive research has been done on the economic consequences of altering the quality of recreational fisheries. To the extent that the supply of recreational fisheries is affected by acidification of lakes in the Park, we will apply the following methodology to estimate associated economic impacts:

1. Project future emissions of $NO_x$ and $SO_x$ emissions in the region from years 1990 to 2020.
2. Estimate deposition levels in the region as a result of projected emissions.
3. Estimate level of lake acidification associated with regulatory scenarios.
4. Quantify economic impacts associated with reducing supply of lakes available for recreational fishing due to acidification.

A flow diagram of this methodology is provided in Exhibit 2, and the models to be used for each step are described in the following discussion.

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[19] Adirondack Park Association. Watershed Protection of the St. Lawrence River Watershed with Special Consideration to Large Wetlands and Large Landownership. Accessed at http://www.apa.state.ny.us/Research/stregis/report/reportpg2.html.

[20] Driscoll, Charles T. et al. May 2003. Chemical Response of Lakes in the Adirondack Region of New York to Declines in Acidic Deposition. Environmental Science and Technology 37(10): 2036-2042.

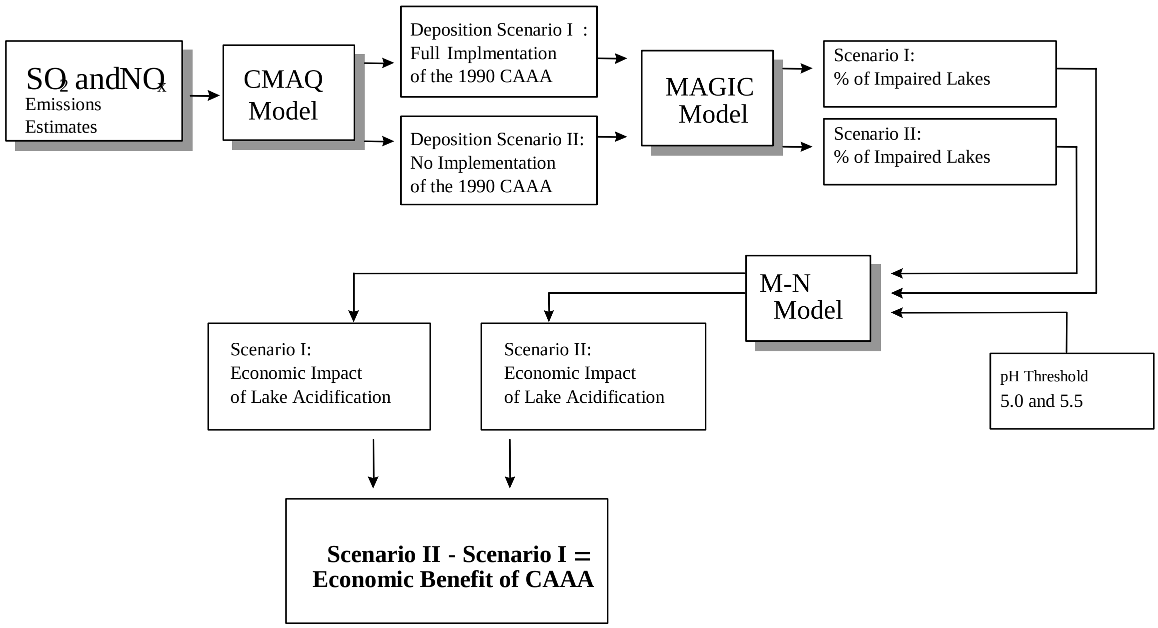
[21] New York State Department of Environmental Conservation (DEC). 1998. Acid Rain and its Effects on Adirondack Lakes. Accessed at http://www.dec.state.ny.us/website/dfwmr/habitat/acidrain.htm.

[22] Driscoll et al., May 2003.

[23] Driscoll, Charles T. et. al. March 2001. Acidic Deposition in the Northeastern United States: Sources and Inputs, Ecosystem Effects, and Management Strategies. BioScience 51(3): 180-198.

[24] Adirondack Lake Survey Corporation. 2005. Acid Rain and the Adirondacks: A Research Summary. Ray Brook, New York.

**EXHIBIT 2: MODEL FOR ESTIMATING THE ECONOMIC BENEFITS OF REDUCED ACIDIFICATION ON ADIRONDACK LAKES**




Source: Industrial Economics, Inc. Memorandum to Brian Heninger, EPA. March 1999. Task 3, WA 4-89: Draft Quantification of the Benefits to Society from Decreased Acidification of Fresh Water Lakes and Streams, 1990-2010.

### Step 1. Project $NO_x$ and $SO_x$ emissions

The first step in the process of estimating the benefits of the CAAA in the Park is the projection of emissions of each pollutant under two scenarios: one in which the CAAA are implemented and one in which they are not. These projections are being forecast as part of the broader Second Prospective analysis, and will be applied in this case study.

### Step 2. Estimate deposition levels

Estimates of emissions will be translated into regional deposition using the Community Multiscale Air Quality (CMAQ) Model, developed by the EPA to model multiple air quality issues, including acid deposition. The CMAQ model may be parameterized temporally and spatially and output is ground level concentrations of pollutants at the 36 kilometer grid level.[25]

### Step 3. Estimate lake acidification levels

To translate future deposition levels into impacts on fisheries, it is first necessary to simulate their transport through the hydrological and terrestrial ecosystems. The Model of Acidification of Groundwater in Catchments (MAGIC) is a peer-reviewed, lumped-parameter model developed to project the long-term effects to surface water caused by acidic deposition.[26] This model has been used extensively by the EPA in analysis of acidification in the Eastern United States and was applied in the first prospective analysis of the benefits of the CAAA.[27]

MAGIC simulates the movement of acidic pollutants through the terrestrial and aquatic ecosystems of the watersheds on which they are deposited. The model includes a mass balance calculation to determine the flux of major ions to and from the system in response to atmospheric inputs, chemical weathering inputs, net uptake in biomass and losses to runoff.[28] MAGIC operates at the catchment scale and is typically simulated in seasonal time steps implemented on decadel or centennial time scales.[29] MAGIC models the response of a sample of lakes in the park that are vulnerable to acidification (as determined by their baseline ANC).

For the purposes of this analysis, the output of MAGIC will be a percentage estimate of the sample lakes in the Park that fall into two categories:

- *Fishable* - Water quality is not deteriorated as to limit recreational fishing.
- *Impaired* - Water quality is deteriorated so as to reduce fish populations and preclude recreational fishing.

We will bin lakes into these categories according to identified pH thresholds. The toxicity of low levels of pH to fish species depends on chemical characteristics of the lake, as well as physiological characteristics of the species themselves. These

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[25] Environmental Protection Agency. 2003. Atmospheric Sciences Modeling: Community Multiscale Air Quality Model. Accessed at http://www.epa.gov/asmdnerl/CMAQ/cmaq_model.html.

[26] Cosby, B.J., et al. 2001. Modelling the Effects of Acid Deposition: Refinements, Adjustments, and Inclusion of Nitrogen Dynamics in the MAGIC Model. Hydrology and Earth System Sciences 5(3):499-517.

[27] Additionally, the biogeochemical PnET-BCG model has been used to simulate the response of soil and surface waters to acid deposition (Chen, Limin and Charles T. Driscoll. 2004. Modeling the Response of Soil and Surface Waters in the Adirondack and Catskill Regions of New York to Changes in Atmospheric Deposition and Historical Land Disturbance. Atmospheric Environment 38:4099-4109). This model was developed to evaluate the effects of atmospheric deposition and land disturbances on soil and surface waters in northern forest ecosystems. While the MAGIC model has been applied more broadly for this purpose, and has most recently been applied by the EPA (NAPAP. 2005. National Acid Precipitation Assessment Program Report to Congress: An Integrated Assessment. Washington, DC.), it may be possible to apply both models to examine sensitivity of results.

[28] Cosby, B.J. et. al. 1985. Time Scales of Catchment Acidification: A Quantitative Model for Estimating Freshwater Acidification.

[29] Wright, Richard F. and Bill J. Cosby. Model of Acidification in Groundwater Catchments. General Model Information from the Register of Ecological Models (REM). Accessed at http://eco.wiz.uni-kassel.de/model_db/mdb/magic.html.

characteristics vary by site within the Park region. Generally, Adirondack fish species prefer neutral or slightly acid conditions and do not tolerate low pHs well.

Many scientific studies, both laboratory and field, have attempted to identify critical acidification thresholds below which a fish species is compromised or cannot survive. Thresholds of pH below which specific fish species are expected to be lost are typically given as ranges of pH values. The EPA's 1990 National Acid Precipitation Assessment Program (NAPAP) Integrated Report suggested that most fish species are lost as pH falls below 4.5 to 5.5. According to more recent research specifically on Adirondack fish and acidity, pH of 5.5 is a stress point beyond which most species do not function well, and below a pH of 5.0 most species cannot survive.[30] In order to account for the uncertainty regarding the varying sensitivities to acidity of the diverse fish species present in Adirondack lakes, and the uncertainty regarding at what pH the effects on fish population are perceptible to anglers, we will apply these two alternative acidification thresholds to define fishable and impaired lakes: 5.5 and 5.0.

As mentioned above, MAGIC models the response of only a subset of lakes in the park that are considered vulnerable to acidification, defined as having an ANC of less than 400 microequivalents per liter (µeq/L). This is because intensive data collection is required to simulate the complex hydrological, biological, and chemical dynamics of the lakes. The MAGIC model applied in the First Prospective analysis involved a sample of 33 lakes considered to be representative of lakes in the region that are considered vulnerable to acidification (have an ANC of less than 400 (µeq/L). We are aware that since the First Perspective, the sample of lakes for which data are available is significantly expanded, and that there is an ongoing effort to continue to expand this dataset. For the Second Prospective study, we will work with Jack Cosby, developer of the MAGIC model, to determine the sample of lakes for which data may be available to run MAGIC, and which are considered representative of the full population of acid sensitive lakes so that results may be extrapolated.

### Step 4. Quantify economic impacts of lake acidification

The output of MAGIC will serve as an input to a linked economic model. While many models exist to describe the economic impacts of changes in water quality on recreational fishing, only one model, developed by Montgomery and Needelman, specifically covers the Adirondack region and accounts for effects of water quality changes on multiple fisheries within a system.[31] The Montgomery and Needelman model applies a random utility model (RUM) to account for fishing site choices made by recreational fishers based on attributes of sites.

The economic model controls for external factors contributing to site choice for fishing, such as geographic location of a lake. The model then estimates the full value of the available set of lakes to each New York resident. It then predicts changes in consumer surplus associated with limiting the sites available (e.g., conversion of lakes from "fishable", to "impaired" as determined by MAGIC and the pH thresholds identified above) for fishing due to deteriorating water quality associated with acidification.

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[30] Adirondack Lake Survey Corporation. 2005. Acid Rain and the Adirondacks: A Research Summary. Ray Brook, New York. Pg. 76.

[31] Montgomery, M. and M. Needelman. 1997. The Welfare Effects of Toxic Contamination in Freshwater Fish. Land Economics 73(2):211-223.

Because the model assumes only two states for lakes, fishable or impaired (complete absence of fishing), the identified pH thresholds of 5.5 and 5.0 represent levels below which game fishes will no longer occur in adequate numbers to be amenable to recreational fishing as described above.

The difference in economic welfare values between the value of the lakes available in the "with CAAA" scenario and the "without CAAA" scenario represents the benefits to recreational fishing in the Adirondack region associated with the CAAA. The economic benefits estimated represent New York State residents' willingness to pay to prevent lake acidification.

#### NEXT STEPS

1. For this analysis, we will research the feasibility of applying existing MAGIC runs. MAGIC is calibrated to a subset of lakes in the Park to simulate the response of these systems to the projected levels of atmospheric deposition beginning in year 1990. If simulations exist for deposition estimates in the region for the EPA that are close to the scenarios being modeled (full implementation, and no implementation, of the Clean Air Act Amendments), it may not be necessary to fund additional runs of this model.
2. Economists at IEc experienced in running the Montgomery and Needelman model will apply the model for this analysis with the oversight of Dr. Mark Montgomery.

#### CAVEATS

This analysis captures the impacts of lake acidification only on recreational fishing. Other ecological attributes may be affected, for example non-game species may be affected or there may be changes in species assemblages (transition to more acid-resistant species), but limited information is available to isolate and quantify the economic impacts of these types of ecological effects.

This analysis does not capture ecological impacts of episodic acidification of surface waters, as may occur during snow melt periods. Such episodes have been observed, for example, in Buck Creek within the Park as part of the EPA's Episodic Response Project (ERP). The ERP was undertaken to monitor acidic events and subsequent mortality of brook trout and blacknose dace in the Adirondack and Catskill mountains and the Appalachian Plateau.[32] Description of these effects can be included in a qualitative discussion of the impacts of acidification in the Adirondacks.

In terms of the ecological modeling, we recognize that pH is not the only factor that determines fish species survival and abundance. Other variables, such as aluminum and calcium concentrations, also affect water chemistry. Concentrations of calcium and magnesium in rain water that help neutralize sulfates have also been decreasing in regional precipitation.[33] These decreases in basic compounds, concurrent with the reductions in sulfates and nitrates, may offset the benefits of reductions in these

[32] Driscoll, Charles T. et. al. March 2001. Acidic Deposition in the Northeastern United States: Sources and Inputs, Ecosystem Effects, and Management Strategies. BioScience 51(3): 180-198.

[33] New York State Department of Environmental Conservation (DEC). 1998. Acid Rain and its Effects on Adirondack Lakes. Accessed at http://www.dec.state.ny.us/website/dfwmr/habitat/acidrain.htm.

compounds.  We do not at this time, however, have an ecological model that accounts for these complex chemical interactions.

Additionally, the Montgomery-Needelman model only accounts for two categories of lake, where game species are abundant or where recreational fishing is infeasible.  In fact, the full scale of acidification may result in many and varied conditions in between these two extremes.  Another simplifying assumption embedded in the Montgomery-Needelman model is that only New York State residents value the Adirondack fisheries on a daily basis.  The model does not account for out-of-state trips for the purpose of fishing, or multi-day fishing trips.

## III. ADIRONDACK CASE STUDY: TIMBER INDUSTRY ANALYSIS

This section describes and outlines the case study of the effects of the CAAA on the timber industry in the Park.  First, general background information about the regional timber industry is highlighted to provide context for the forthcoming quantitative analysis.  Second, the methodological approach to quantify the impacts of pollutants regulated by the CAAA on timber growth is described, including information on the specific models that will be applied to translate ozone, nitrogen, and sulfur deposition on forests into economic impacts on the commercial forestry industry.

## ADIRONDACK TIMBER INDUSTRY PROFILE

Approximately 5.4 million acres of the Park are forested. Timber logging, however, is not permitted on State-owned lands except through specific easements.[34,35,36] The timber available for commercial harvest is on private lands, which constitute approximately half of the Park.[37]

Two and a half percent of New York's 15,781,600 acres of forestland is harvested annually. By comparison, the rate of timber harvesting within the Park is about double that figure, or five percent annually.[38] Almost two thirds of the forest within the Park is considered hardwood forest, dominated by maple-beech-birch and red spruce-balsam fir forests. On a biomass volume basis, the most prevalent species are ranked accordingly: Sugar Maple, Red Maple, American Beech, Eastern White Pine, Eastern Hemlock, Spruce, Yellow Birch, Aspen, and Black Cherry.[39,40]

### Timber Harvesting in Adirondack Park

Timber harvest statistics are reported according to the uses of the timber. In general, harvested timber is categorized either as log, roundwood, or wood chips. Logs are used by sawmills to produce lumber, and are often referred to as "saw-logs". Roundwood and wood chips are used by the pulp and paper industry and for firewood. The amount of wood harvested is either reported in board feet (BF) (when it is a saw log), by cord (when it is roundwood), or by green tons (when it is wood pulp or wood chips).[41,42]

Timber harvesting has been a traditional endeavor in the Park and the region, along with the associated wood product industries.[43] While institutions own a large proportion of privately owned land within the park, individuals or small companies perform the majority of the timber harvesting. Because there is no formal registration or licensing of timber harvesters, exact statistics regarding timber harvests are not collected. Rather, estimations of timber harvests are made from self-reported surveys.[44] As of 2006, two pulp and paper mills are in operation in the Adirondack region, only one of which is located within the blue line and three wood-fired power plants are in operation nearby (two in NY, and one in VT).[45] Pulp and paper industries and wood-powered utility

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34 Bureau of Land Management (BLM). 2006. Adirondack Mountains: Phisiographic Area 26 Executive Summary. Website accessed August 31, 2006 at: http://www.blm.gov/wildlife/pl_26sum.htm.

35 Erman S. 2006. Adirondack Park Agency. Personal Communication. August 31, 2006.

36 Crawford, S. 2006. New York State Department of Environmental Conservation, Forest Products Utilization Program. Personal Communication. September 8, 2006.

37 According to the 1993 Forest Inventory Analysis produced by the USDA Forest Service (the most recent data available), approximately 33 percent of New York's forests and 41 percent of New York's timber harvest are contained within the 11 counties that, in part or whole, officially comprise the Adirondack Park Region (United States Department of Agriculture-Forest Service (USDA FS). 1993. Northeastern Forest Inventory and Analysis. Statewide Results. 1993. Available at: http://www.fs.fed.us/ne/fia/states/ny/1993.html.). However, the Adirondack Park comprises 14 counties in whole or in part (Gould and Holmes, 2006). This inventory analysis therefore considers timber activity only in portions of the Park.

38 Crawford, 2006.

39 BLM, 2006.

40 Gould and Holmes, 2006.

41 One thousand board feet is abbreviated as "MBF", and one million board feet is abbreviated "MMBF". One board foot is a section of lumber that is one inch by 12 inches by 12 inches. (Smartwood Program. 2003. Forest Management Public Summary for: New York State Department of Environmental Conservation. Smartwood Program, New York, NY. 2003.)

42 A cord of wood is a stack of roundwood that measures four feet by four feet by eight feet (128 cubic feet). (Smartwood Program, 2003.)

43 Gould and Holmes, 2006.

44 Crawford, 2006.

45 Ermin, 2006.

companies historically have dominated the local consumption of timber. These industries demand high volumes of wood chips, as opposed to high quality logs. While the elevation, climate, management history and ownership of the forests within the park are important factors, it is the proximity of these industries that has historically dictated that the quality of the wood that is harvested in the Park is generally lower that that of forests in other parts of New York State.[46]

While 80 percent of the pulpwood and fuel chips produced in New York State in 2004 were produced in Adirondack Park, more valuable saw-timber is increasingly harvested.[47] Based on USDA Forest service data from 1993, approximately 41 percent of the timber harvested for saw-logs was harvested from the 11 Adirondack counties that together form the three official Forest Inventory Analysis Adirondack Regions.[48,49] This proportion can be used as a baseline estimate for the proportion of timber harvesting in New York State that takes place within Adirondack Park.[50]

### Value of Timber Harvests

Exact estimates of the quantities and value of timber harvested within the Park are difficult to calculate because neither State agencies nor the Federal government record statistics on timber harvesting at the Park level. To provide context for the timber impacts analysis, survey data and information collected at the county level are used to estimate the quantity and value of timber harvested within the Park.

The type and estimated amount and value of wood harvested annually within the Park are presented in Exhibit 3. Based on estimates for annual stumpage revenue generated in New York State, and the proportion of forestland located within the official Adirondack regions outlined by the U.S. Department of Agriculture Forest Service (USDA FS), estimated annual stumpage revenue from saw timber harvesting within the Park is greater than $100 million.[51] Current average stumpage prices paid for saw timber in New York for the most common tree species in the park, and historical stock values from 1993 are listed in Exhibit 4.

Three hundred thousand to 500,000 green tons of fuel wood chips are harvested from the Park annually, out of an estimated 1,840,000 green tons of wood harvested for the pulp and paper and wood-burning utilities industries in 2004.[52] The estimated annual revenue generated from the sale of fuel wood chips from the Park is over $5 million.[53] The

[46] Crawford, 2006.

[47] Ibid.

[48] Those same 11 counties contained approximately 33 percent of the New York's timber volume.

[49] USDA FS, 1993.

[50] Crawford, 2006.

[51] Calculated based on proportion of NY timber harvest located within 11 counties contained within the three official FIA Adirondack regions (41.2 percent) and the estimate of stumpage revenue of $250 million for NY State in 2004 (USDA FS, 1993.; North East State Foresters Association (NESFA). 2004. The Economic Importance of New York's Forests. December, 2004.).

[52] Total pulpwood and woodchip harvest figure was calculated from an estimation of 80 percent of wood chips generated in New York originating from within the Park, and the value for the total quantity of wood chips in New York (2.3 million green tons) (Crawford, 2006.; NESFA, 2004.). The estimate for the quantity of fuel wood chips is from an estimate of 373,118 green tons of the same produced in 2002 in New York, revised upwardly for increased production over time and the estiation of 80 percent of wood chips generated in New York originating from within the Park (NESFA, 2004.; Crawford, 2006.)

[53] This number was calculated from an estimation of 80 percent of wood chips generated in New York originating from within the Park, and the value for the total revenue generated from sales of wood chips in New York (Crawford, 2006.; NESFA, 2004.).

average price paid for fuel wood chips in 2004 was $18/ton.[54] The revenue generated from roundwood harvested for the pulp and paper industry is unknown, but can be roughly estimated, based on the proportion purchased and the price per ton listed above, at approximately $25 million.[55] As can be seen in Exhibit 5, the total value of the timber industry in the Park is likely higher than $130 million.

[54] NESFA, 2004.

[55] Of the 2.3 million green tons of pulpwood and wood chips harvested, 35 percent was purchased for fuel wood chips, and 65 percent was purchased for the pulp and paper industry (New York State Department of Environmental Conservation (NYS DEC). 2004. New York State Industrial Timber Harvest Production and Consumption Report-2004. Forest Products Utilization Program. Albany, NY.).

**EXHIBIT 3. TIMBER INDUSTRY STATISTICS FOR ADIRONDACK PARK**

| STATISTIC | ESTIMATE |
|---|---|
| Estimated quantity of timber harvested for saw-log production[a] | 334,132,000 BF |
| Estimated stumpage revenue generated[b] | $103,000,000 |
| Estimated quantity of timber harvested for pulpwood and fuel chips[c] | 1,840,000 Green Tons |
| Estimated revenue generated from the sale of fuel chips only[d] | ~$5,000,000 |
| Estimated acres of forest harvested annually[e] | 150,353 Acres |

[a]=Calculated based on proportion of NY timber harvest located within 11 counties contained within the three official FIA Adirondack regions (41.2 percent) and the saw timber harvest volume estimate provided of 811 million BF for NY State in 2004 (United States Department of Agriculture- Forest Service (USDA FS). 1993. Northeastern Forest Inventory and Analysis. Statewide Results. 1993. Available at: http://www.fs.fed.us/ne/fia/states/ny/1993.html.; New York State Department of Environmental Conservation (NYS DEC). 2004. New York State Industrial Timber Harvest Production and Consumption Report-2004. Forest Products Utilization Program. Albany, NY.).

[b]=Calculated based on proportion of NY timber harvest located within 11 counties contained within the three official FIA Adirondack regions (41.2 percent) and the estimate of stumpage revenue of $250 million for NY State in 2004 (USDA FS, 1993.; North East State Foresters Association (NESFA). 2004. The Economic Importance of New York's Forests. December, 2004.).

[c]=Calculated based on proportion of NY pulpwood and wood chips produced in Adirondack Park (80 percent) and the estimate for NY State in 2004 (2.3 million green tons) (Crawford, S. 2006. New York State Department of Environmental Conservation, Forest Products Utilization Program. Personal Communication. September 8, 2006.; NYS DEC, 2004.)

[d]=Calculated based on proportion of pulpwood and wood chips produced in Adirondack Park (80 percent) and the estimate of $6.7 million provided for NY State in 2004 (Crawford, 2006.; NESFA, 2004.).

[e]= Calculated based on acres of land located within Adirondack Park susceptible to timber harvests (approximately three million acres) and the estimate of the proportion of forests harvested annually in Adirondack Park (five percent) (Adirondack Park Agency (APA). 2003. Acreage by County and Land Use Classification. March, 2003. Website accessed August 31, 2006 at: http://www.apa.state.ny.us/gis/colc0303.htm.; Crawford, 2006.).

**EXHIBIT 4. ESTIMATED BASE PRICE FOR MOST COMMON TREE SPECIES IN ADIRONDACK PARK IN 2006**

| TREE SPECIES | ESTIMATED PRICE IN USD[A] FOR SAW TIMBER PER MBF | VOLUME OF TREES (MMBF)[B] | VOLUME OF TREES (THOUSANDS OF TREES)[C] |
|---|---|---|---|
| Sugar Maple | 540 | 3,826 | 116,661 |
| Red Maple | 175 | 2,361 | 136,966 |
| American Beech | 40 | 1,446 | 73,915 |
| Eastern White Pine | 55 | 2,750 | 40,473 |
| Eastern Hemlock | 40 | 1,914 | 54,753 |
| Spruce | 55 | 1,050 | 39,412 |
| Yellow Birch | 105 | 1,028 | 36,412 |
| Aspen | 55 | 1,133 | 31,267 |
| Black Cherry | 900 | 1,156 | 29,438 |

[A] (Office of Real Property Services (NYS ORPS). 2006. Forest Region 6 Stumpage Price Schedule. Website accessed on September 11, 2006 available at: http://www.orps.state.ny.us/assessor/resources/forestry/stumpreg6.cfm.)

B Calculated from Tables 87, 99, 111 of the 1993 FIA (United States Department of Agriculture- Forest Service (USDA FS). 1993. Northeastern Forest Inventory and Analysis. Statewide Results. 1993. Available at: http://www.fs.fed.us/ne/fia/states/ny/1993.html.).

[C] Calculated from Tables 80, 92, 104 of the 1993 FIA (USDA FS, 1993.).

### EXHIBIT 5. ESTIMATED VALUE OF TIMBER INDUSTRY IN ADIRONDACK PARK

| HARVEST TYPE | ESTIMATED VOLUME | ESTIMATED VALUE |
|---|---|---|
| Saw logs | 334,132,000 BF[a] (~1,700,000 Green Tons)[b] | $103,000,000[c] |
| Fuel Wood Chips | 300,000 to 500,000 Green Tons[d] | ~$5,000,000[e] |
| Wood for Pulp and Paper | 1,300,000 to 1,500,000 Green Tons[f] | ~$25,000,000[g] |
| Total | ~3,500,000 Green Tons | >$130,000,000 |

[a]=Calculated based on proportion of NY timber harvest located within 11 counties contained within the three official FIA Adirondack regions (41.2 percent) and the saw timber harvest volume estimate provided of 811 million BF for NY State in 2004 (United States Department of Agriculture- Forest Service (USDA FS). 1993. Northeastern Forest Inventory and Analysis. Statewide Results. 1993. Available at: http://www.fs.fed.us/ne/fia/states/ny/1993.html.; New York State Department of Environmental Conservation (NYS DEC). 2004. New York State Industrial Timber Harvest Production and Consumption Report-2004. Forest Products Utilization Program. Albany, NY.).

[b] Conversion factor of 1000 BF = 5 Green Tons (United States Department of Agriculture- Forest Service (USDA FS). 1993. Northeastern Forest Inventory and Analysis. Statewide Results. 1993. Available at: http://www.fs.fed.us/ne/fia/states/ny/1993.html.)

[c] Calculated based on proportion of NY timber harvest located within 11 counties contained within the three official FIA Adirondack regions (41.2 percent) and the estimate of stumpage revenue of $250 million for NY State in 2004 (USDA FS, 1993.; North East State Foresters Association (NESFA). 2004. The Economic Importance of New York's Forests. December, 2004.).

[d]Total pulpwood and woodchip harvest figure was calculated from an estimation of 80 percent of wood chips generated in New York originating from within the Park, and the value for the total quantity of wood chips in New York (2.3 million green tons) (Crawford, S. 2006. New York State Department of Environmental Conservation, Forest Products Utilization Program. Personal Communication. September 8, 2006.; NESFA, 2004.). The estimate for the quantity of fuel wood chips is from an estimate of 373,118 green tons of the same produced in 2002 in New York, revised upwardly for increased production over time and the estiation of 80 percent of wood chips generated in New York originating from within the Park (NESFA, 2004.; Crawford, 2006.)

[e]Calculated based on proportion of pulpwood and wood chips produced in Adirondack Park (80 percent) and the estimate of $6.7 million provided for NY State in 2004 (Crawford, 2006.; NESFA, 2004.).

[f]Calculated by subtracting the value for fuel and woodchips (note d above) from the total estimate of pulpwood and woodchips produced in the park, which is alculated based on proportion of NY pulpwood and wood chips produced in Adirondack Park (80 percent) and the estimate for NY State in 2004 (2.3 million green tons) (Crawford, 2006.; NYS DEC, 2004.)

[g]Calculated based on note f above and value of $18/green ton given in (NESFA, 2004.)

#### Timber Harvesting Caveats and Uncertainty

Harvest and the removal of timber has increased by nearly 90 percent since 1968 within the Park, and Adirondack forests have increased in volume and acres steadily since 1900.[56] As of 1993, the ratio of growth of saw timber to removal was three to one, demonstrating that volume may not be a limiting factor in silviculture production.[57]

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[56] BLM, 2006.

[57] USDA FS, 1993.

Much of the harvesting of timber within the Park is harvesting of existing stands in a sustainable manner. In addition, one important caveat regarding timber harvesting within the Adirondack region is the effect of storms and forest damage. The ice storm of 1998, for example, caused the focus of timber harvesting to focus on damaged trees for several years after the storm in lieu of harvesting at other planned sites.[58] Consequently, the effect of changes in growth rates of trees on timber harvesting is a function of multiple variables and may be offset by external factors.

This valuation of the timber industry does not include any value attributable to tourism or the value the forest ecosystem provides as a habitat. Nor is the value of secondary manufacturing and associated wood products included in this estimation of the value of the timber industry. The Adirondack region has a long history of woodworking, furniture making, and boat building, which is dependent upon the local supply of wood, and wood products companies provide a significant source of employment.[59] The pulp and paper industry alone is directly responsible for over 700 jobs within the Park.[60] These economic benefits may or may not be affected by changes in the productivity of the timber industry.

### ECOLOGICAL EFFECTS OF OZONE, NITROGEN, AND SULFUR ON FORESTS

The ecological effects of ozone on plant growth are direct and have been well documented. The effects of nitrogen and sulfur deposition on plants are indirect and less well documented. Toxic effects are summarized as follows:

- Ozone exerts a toxic effect on the photosynthetic mechanism employed by plants. The route of exposure is via the leaf stomata. Because the opening and closing of leaf stomata is dependent upon many factors, but principally water availability, drought conditions may have a protective effect against the toxic effects of ozone exposure in plants. Concentration-response curves have been established for several tree species. These curves quantify the relationship between ambient ozone concentration and tree biomass.[61] The effect of ozone on adult trees is still debated however.[62]

- Nitrogen and sulfur deposition has widely been associated with the acidification of soils. Though nitrogen is a nutrient for plant growth, and increased nitrogen deposition can have a fertilization effect, soil acidity caused by the deposition of nitrogen and sulfur chemical species ($SO_2$ and $NO_x$) can cause the leaching of other cationic nutrients from soils. Specifically, the leaching of calcium and magnesium from soils due to acidic soil conditions have been associated with decreased plant growth and dieback of trees; however, this relationship is complex and not completely elucidated.[63] There is some evidence to suggest that calcium / aluminum ratios below one, and soil base saturation below 20 percent are

[58] Crawford, 2006.

[59] Gould and Holmes, 2006.

[60] Ermin, 2006.

[61] Abt Associates, Inc. 2006. Technical Report on Ozone Exposure, Risk, and Impact Assessments for Vegetation: Draft Report. Abt Associates, Inc., Bethesda, MD. July 13, 2006.

[62] Chappelka, A.H. and L.J. Samuelson. 2006. Ambient Ozone Effects on Forest Trees of the Eastern United States: A Review. New Phytologist 139(1):91-108.

[63] Driscoll, C.T. et. al. 2001. Acidic Deposition in the Northeastern United States: Sources and Inputs, Ecosystem Effects, and Management Strategies. BioScience 51(3)180-198.

thresholds for toxicity to forest vegetation, but these thresholds have not been validated.[64] Also, variability in the baseline nutrient status of the soils, the complex nature of forest ecosystems, and the contributions of multiple stressors (e.g. disease, insect infestation) complicate prediction of the effects of acid deposition. However, nitrogen fertilization can increase stomatal conductance, and thereby expose vegetation to the deleterious and toxic effects of ozone.

## METHODOLOGICAL APPROACH FOR TIMBER ANALYSIS

To assess the effect of ozone, nitrogen, and sulfur air pollutants on timber growth in the Park, a step-wise methodology will be employed. Our approach is as follows:

1. Project future concentrations of ground level ambient ozone concentration and nitrogen and sulfur deposition in the Park.
2. Quantify the effects of these pollutant levels on the growth of timber,
3. Determine the economic impacts of any observed changes in timber growth.

The remainder of this section of the white paper details this stepwise methodology.

### Step 1. Obtain estimates of air pollutant deposition

As described in the previous section, ground level estimates of ozone deposition will be estimated by translating the EPA's emissions estimates into deposition estimates applying the CMAQ model.

### Step 2. Quantify effect of pollutant scenarios on timber growth

Two models, which have been used in both previous reports by EPA and in peer-reviewed journals to assess the effects of air pollutants on tree growth, are the PnET model and the TREGRO-ZELIG model.[65,66] Each of these models has benefits and drawbacks regarding how they calculate tree growth and the effects of air pollutants on that growth. A basic overview of their functioning and benefits and drawbacks is discussed below.

- **PnET-II Model -** The original PnET model was first developed in 1995 at the University of New Hampshire and now comprises three distinct models: PnET-Day, PnET-II, and PnET-CN. Each of the models incorporates site-specific information about climate, and generalized information about tree growth to estimate forest stand level growth. Growth is directly dependent upon carbon, water, and nitrogen availability, is modeled through monthly time-step iterations, and is estimated by taking into account net photosynthesis through a multi-layered

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[64] Cronan, C.S. 1990. Relationships between Aqueous Aluminum and Acidic Deposition in Forested Watersheds of North America and Northern Europe. Environmental Science and Technology 24:1100-1105.; Gbondo-Tugbawa, S.S. and C.T. Driscoll. 2002. Evaluation of the effects of future controls on sulfur dioxide and nitrogen oxide emissions on the acid-base status of a northern forest ecosystem. Atmospheric Environment 36:1631-1643.

[65] Environmental Protection Agency (EPA). 1999. The Benefits and Costs of the Clean Air Act 1990 to 2010. EPA Report to Congress. Office of Air and Radiation, Office of Policy. November 1999.; Abt Associates, Inc., 2006.

[66] Ollinger, S.V., J.D. Aber, P.B. Reich, and R.J. Freuder. 2002. Interactive effects of nitrogen deposition, tropospheric ozone, elevated $CO_2$ and land use history on the carbon dynamics of northern hardwood forests. Global Change Biology 8:545-562.; Weinstein, D.A., B. Gollands, and W.A. Retzlaff. 2001. The Effects of Ozone on a Lower Slope Forest of the Great Smoky Mountain National Park: Simulations Linking an Individual Tree model to a Stand Model. Forest Science 47(1):29-42.

canopy.[67,68] Each of the models can be parameterized for single-species or multiple-species stands, though a detailed understanding of the model and its relevant input parameters is necessary. Parameters for Northern Hardwood Forests have been published.[69] PnET-CN, which takes into consideration carbon and nitrogen cycles, is the model most appropriate of the three for determining the effects of air pollutants on tree growth. In this model, the toxic effect of ozone can be explicitly included, and ambient ozone concentrations are modeled to have a linear negative effect on photosynthesis.

Because growth is dependent upon foliar nitrogen content, allowing changes in the available nitrogen pools allows for the inclusion of nitrogen deposition rates in the model. Because fertilization rates can affect stomatal conductance, there is potential for the model to capture increased toxicity of ozone due to nitrogen deposition. Unfortunately, because the relationship between soil acidity associated with nitrogen and sulfur deposition is not clearly understood, the potential effect of nutrient leaching due to acidic deposition is not explicitly accounted for in the model.

- **TREGRO-ZELIG Model -** The TREGRO-ZELIG model is, in fact, two models that were joined together. The TREGRO model predicts individual tree growth, and the ZELIG model applies estimated changes in individual tree growth to model the competitive interactions of mature trees during succession to estimate species growth and success within a stand.[70]

  The TREGRO model was developed by Dr. David Weinstein at Cornell University, and models the physiological changes of a single tree in response to input environmental parameters. Like the PnET model, it models growth according to changes in carbon, water and nutrient availability. Whereas the PnET model (except for the PnET-Day model) utilizes monthly time-step iterations, the TREGRO model uses daily time-step iterations, and is usually run for a period of about three years. The TREGRO model was originally designed to model seedling and sapling growth rates in ozone chambers. Consequently, parameters are often validated using site-specific empirical data.[71]

  The TREGRO model must be parameterized for each tree species modeled. Currently, parameters have been calculated for the following tree species relevant to Adirondack Park: Sugar Maple, Red Maple, Spruce, Aspen, and Black Cherry. American Beech and Eastern Eastern Hemlock have not been parameterized, but because of their slow growth, are not expected to be affected by ozone. Yellow Birch and White Pine have not been parameterized to date, and consequently represent potential shortfalls in the completeness of any potential analyses.[72]

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[67] Complex Systems Research Center (PnET). 2006. About PnET. Institute for the Study of Earth, Oceans and Space at the University of New Hampshire. Official Website accessed September 15, 2006 available at: http://www.pnet.sr.unh.edu/.

[68] Ollinger et. al., 2002.

[69] Ibid.

[70] Weinstein et. al., 2001.

[71] Weinstein, D.A. 2006. Personal Communication September 13, 2006.

[72] Ibid.

The toxic effects of ozone are incorporated into the TREGRO-ZELIG model in three ways. First, ozone is modeled to exhibit a negative effect on photosynthesis, similar to how such toxicity is modeled in the PnET model. Decreased growth in the TREGRO model leads to decreased height and crown expansion. Second, decreased growth leads to decreased leaf density. The effects of decreases in these factors are accounted for in the ZELIG model, which models the allocation of sunlight through the canopy. Finally, the ZELIG model also models a decrease in fine root mass associated with increased ozone concentrations, which lead to a decrease in drought tolerance, and therefore survival, of the tree.[73]

Practically, the effect of ozone is included in the TREGRO model as a decreased growth factor. This factor adjusts growth in each iteration of the model. The TREGRO model does not have the ability to incorporate changes in nutrient levels over time, as the PnET model does. Consequently, it does not model the simultaneous effect of both nitrogen and sulfur deposition and ozone on tree growth. But because the TREGRO model incorporates a simple growth factor to account for the effect of ozone, if the "effect" of nutrient leaching or acid deposition on tree growth were known, such an effect could be incorporated into the TREGRO model simply by adjusting this growth factor. However, this effect would have to be viewed in isolation of any effect of ozone as multiple growth factors cannot be included, and the cumulative effect of ozone and nitrogen and sulfur deposition is not known.[74]

Given the inherent strengths and weaknesses of each of the tree growth models, it is likely that the effect of ozone on timber growth at the scale of the Adirondack Park could be estimated using either model. Because the effect of acidic deposition on tree growth is poorly understood, and is not clearly outlined in the scientific literature, neither model has the capability to model any effects of this phenomenon. The PnET-CN model, however, has the added functionality of incorporating nitrogen deposition. In addition, because it is anticipated that PnET will be used for the larger nation-wide analysis, as it was in the First Prospective, there is a possibility that a subset of results from the larger analysis may be able to be used for the case study without additional model runs. For these reason, and because the TREGRO-ZELIG model may not have complete parameterization of the relevant tree species located within the Park, the PnET model is the model that we propose to use for this step in the analysis. In order to effectively use the PnET model, we believe that Dr. Scott Ollinger, an author of the PnET model, would need to be retained to either run the model, or to serve as a consultant. Dr. Ollinger played a similar role in the First Prospective analysis.

#### Step 3. Quantify Economic Impacts of Changes in Timber Growth

While the TAMM model was used to quantify the economic effects of changes in timber growth due to ozone in the First Prospective analysis (The Benefits and Costs of the Clean Air Act 1990 to 2010, Appendix E), this model is not useful for assessing economic impacts at the scale of this case study.[75] Dr. Richard Haynes of the United States Forest Service, the primary author of the TAMM model, advised us on the use of a spreadsheet-based model to quantify economic impacts of changes in timber growth at scales relevant to this case study.

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73 Weinstein et. al., 2001.

74 Weinstein, 2006.

75 EPA, 1999.

The spreadsheet-based model will translate changes in timber growth into changes in forest inventories; inventories may then be tracked over time iteratively, taking into account increases in inventory due to growth and decreases in inventory due to harvest. Over time, any decreased growth rate leads to decreased forest inventory, and any decreased forest inventory is monetized according to stumpage prices.[76] Each iteration of the model will likely represent a year, and the model will likely be run for a period of 50 to 75 years.

Inputs to this model will be the following:

- Base forest inventory (either in acres, or in cubic feet per acre per year)
- Annual "effect" of pollutant expressed as a decrease in net growth rate (either as a percentage loss of acres or a numerical loss in cubic feet per acre per year)[77]
- Harvest rate (either as a percentage loss or a numerical loss in cubic feet per acre per year)
- Stumpage price conversion[78]
- Stumpage price projection
- Discount rate

Because forests are harvested in a sustainable manner in the Park, and clear cutting is not common, the model will not take into account any effect of new plantings. Regeneration of trees will be incorporated into the net growth rate, since growth rate will be expressed on a biomass or acreage basis.

Each iteration of the model (iterated across year cells in the spreadsheet) will track forest inventory according to the following expression:

$$Inventory = f(inventory\ from\ previous\ year + net\ growth\ rate - harvest)$$

Stumpage price conversions for yearly iterations will be calculated from the original stumpage price conversion and the stumpage price projection. Each yearly inventory will then be monetized according to the following expression:

$$Value\ of\ timber = \Sigma(inventory - inventory\ from\ previous\ year) * stumpage\ price\ conversion)$$

Finally, the value of timber will be discounted over the length of time that the model is run.[79]

## IV. TOTAL VALUE OF NATURAL RESOURCE IMPROVEMENTS IN THE ADIRONDACK REGION

This section summarizes the intersections of Resources for the Future's (RFF's) recent research regarding the total value of resource improvements in the Adirondack region.

[76] Haynes, R. 2006. U.S. Department of Agriculture- Forest Service. Personal Communication. September 8, 2006.

[77] Net growth rate is the growth rate adjusted for mortality.

[78] This is a conversion factor which monetizes forest inventory according to assumptions about quantities of individual tree species harvested.

[79] This analysis will apply a discount rate of five percent to reflect the social rate of time preference as established for the broader Second Prospective analysis.

RFF provided a report to the project team  detailing the applicability of their results to the case study of the ecological benefits of the CAAA in the Adirondacks.

Specifically, RFF's research focuses on quantitative analysis of total value associated with reduced lake acidification and other ecological improvements associated with reductions in acid precursors. This research was completed under the auspices of EPA/OAR/CAMD and is described in the September 2004 RFF Report *Valuation of Natural Resource Improvements in the Adirondacks*, authored by Spencer Banzhaf, Dallas Burtraw, David Evans, and Alan Krupnick. A shorter version of this study was published in *Land Economics*, a peer-reviewed journal, in August of 2006.

Based on a contingent valuation survey of New York residents, the RFF study quantifies the total economic value of expected ecological improvements in the park from forthcoming policies. The preferred estimates of the mean household willingness to pay range from $48 to $107 annually, implying total statewide benefits ranging from $336 to $749 million annually. Alternative assumptions about ecological changes yield benefits up to $1.1 billion.  Importantly, the RFF study estimates a "total value", incorporating both use and non-use dimensions of the Park resources.  It is not possible to isolate the non-use component of the total value with the existing study.  It is therefore important to note that the total value estimates, even if refined to be relevant the scope of the Second Prospective ecological benefits case study, are not either a perfect subset or superset of the recreational fishing impacts and commercial timber impacts that will be quantified as described above.

RFF reviewed the draft emissions estimation study for the Second Prospective study and compared the scenarios described in this study to those underlying the RFF Adirondacks valuation study.  RFF focused on whether the their research may be applied "as-is", either quantitatively or as a means to estimate in rough terms the magnitude of benefits associated with compliance with CAAA regulations; with modest refinements or adjustments; with more substantial refinements that would require additional funding and research (steps to be outlined in this report); or whether the RFF report is not applicable to the 812 scenarios under any conditions.

The RFF study applies two versions of their survey to bracket the range of uncertainty in the science and to create an opportunity to test the sensitivity of willingness to pay to the scale of ecological improvements; they refer to these as the *base* and a *scope* versions of the survey.  Both the *base* and the *scope* surveys propose an intervention program that would involve improving the health of the lakes by liming them.  Thus, there was no explicit link of the surveys to changes in emissions.

The *base* version of the survey indicates that in 2003 about half of the lakes in the Adirondack region are injured due to acid deposition and said that the program would improve about 20 percent of the lakes, along with minor improvements to birds and forests.  The *scope* version of the survey also indicates that about half the lakes are currently injured and, absent intervention, an additional five percent of the lakes would worsen over time. This survey indicates that 90 percent of the lakes would be improved by the liming program, and also that liming the forests and would improve them to a 99 percent healthy state.

RFF suggests that their willingness-to-pay (WTP) estimates from the base version of their survey can serve as an upper bound to apply to the lake improvements associated with the core scenarios applied in the Second Prospective economic analysis. While past EPA studies have estimated only a two to four percent improvement in lakes as a result of the CAAA,[80] as opposed to RFF's 20 percent, RFF considers there to be less uncertainty about their base case results being applied to the improvement in lakes that encompass the entire period from pre-CAAA to the Clean Air Interstate Rule (CAIR) rule and other regulations as the improvements in lakes from pre-CAAA to post Title IV are larger and may be better represented by a 20 percent improvement. The Project Team will provide RFF the results of MAGIC runs (percentage of affected lakes) that will be applied to the ecological benefits case study in the Adirondacks, along with our relevant assumptions so RFF can analyze them to determine the suitability of our WTP estimates.

While the WTP estimates for the base version of our survey may be appropriate for the Second Prospective Study, RFF cautions that the results of the scope version of their survey (as described in the attachment) are not. The scope version is more optimistic about the degree of improvement than is reasonable or possible under existing regulations or policies. Thus, RFF does not recommend its use.

Caveats associated with using the RFF study results are:

- **The spatial extent of the market.** RFF limits the spatial extent of their analysis to adults living in New York State. Whether these results can be applied outside of New York State is uncertain. RFF found that WTP declined with distance from residence to the Park. This relationship could be used to extrapolate beyond New York State. However, in their judgment, this would be risky as the survey mechanism for eliciting WTP was an increase in state taxes that would go into an Adirondacks fund.
- **The intention of the Second Prospective Study team to compare use values for New Yorkers to the RFF total value estimates.** RFF found that frequent visitors to the Adirondacks tend to have a larger WTP than non-users, when "use" was defined as fairly intense use (over 10 visits per year). However, RFF did not estimate a separate use value. Also, RFF's "commodity" was a 20 percent improvement in lakes of concern. They doubt that the Montgomery-Needelman RUM analysis will be easily comparable. Thus, caution will be called for in comparing any use value estimate to our total value estimates.
- **Time frame of analysis.** RFF's WTP estimates will need to be adjusted for the shortened period of improvement, given an acceptable assumption of a rate of time preference, which in the case of the Second Prospective will be five percent.

[80] USEPA. 1995. Acid Deposition Standard Feasibility Study Report to Congress. EPA 430-R-95-001a.

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## ATTACHMENT: APPLICABILITY OF RFF RESEARCH ON THE TOTAL VALUE OF NATURAL RESOURCE IMPROVEMENTS IN THE ADIRONDACKS TO THE SECOND PROSPECTIVE ECOLOGICAL BENEFITS CASE STUDY

- Alan Krupnick, David Evans, Anna Mische John, and Dallas Burtraw, Resources for the Future

### I. BACKGROUND TO RFF'S ROLE IN THE 812 STUDY

The purpose of this Work Assignment is to continue efforts pursuant to development of the second Section 812 prospective study (called the Second Prospective Study below) which were commenced under various Work Assignments issued under this contract. In particular, this Work Assignment is to initiate the first phase of the ecological benefits analysis component of the second 812 prospective study, consistent with the methodologies proposed in the May 2003 Analytical Blueprint and with the advice conveyed by the 812 Council's Ecological Effects Subcommittee (EES) in their June 2005 report [EPA-COUNCIL-ADV-05-001]. RFF's role is on the second of two tasks under this work assignment: an ecological benefits case study in the Adirondacks area of New York State.

Under this task, RFF is to contribute to an upland ecosystem case study for the Adirondacks. Specifically, RFF's role focuses on quantitative analysis of non-use (a component of total) value associated with reduced lake acidification and other ecological improvements associated with reductions in acid precursors. This research was completed under the auspices of EPA/OAR/CAMD and is described in the September 2004 RFF Report *Valuation of Natural Resource Improvements in the Adirondacks*, authored by Spencer Banzhaf, Dallas Burtraw, David Evans, and Alan Krupnick. A shorter version of this study was published in *Land Economics*, a peer-reviewed journal, in August of this year.

RFF has reviewed the draft emissions estimation study and technical memorandum outlining the development of scenarios for the Second Prospective study. We have compared the scenarios described in this study to those underlying the RFF Adirondacks valuation study and developed recommendations as to the suitability of this research as a case study for the Second Prospective study. We have focused on whether the RFF report can be applied "as-is", either quantitatively or as a means to estimate in rough terms the magnitude of benefits associated with compliance with CAAA regulations; with modest refinements or adjustments (which could be made/included in the report to be provided under this scope of work); with more substantial refinements that would require additional funding and research (steps to be outlined in this report); or whether the RFF report is not applicable to the 812 scenarios under any conditions.

This memo is organized into four sections. The first two (II. and III.) are background to our survey and the underlying science. The next is a section analyzing the applicability of our WTP estimates to the Second Prospective Study, but based entirely on literature we originally used in designing our survey (IV.A). The last section looks at additional literature to make the same judgments (IV.B.).

## II. BACKGROUND TO RFF'S ADIRONDACKS STUDY

For 20 years acid rain in the Adirondack Park has been a central issue in the debate about clean air regulation. Based on a contingent valuation survey of New York residents, the RFF study quantifies the total economic value of expected ecological improvements in the park from forthcoming policies. Our preferred estimates of the mean household willingness to pay range from $48 to $107 annually, implying total statewide benefits ranging from $336 to $749 million annually. Alternative assumptions about ecological changes yield benefits up to $1.1 billion. The instrument passes external scope, sensitivity to bid, and sample selection tests.

This study seeks to fill the gap between ecological and economic estimates within the important context of valuing the benefits of air pollution policies by using the contingent valuation survey approach to estimate the change in the total economic value (the sum of use and nonuse value) to New York State residents that would likely result from an improvement in the Adirondack Park ecosystem through further reductions in air pollution. The survey was administered both on the Internet and via mail, providing a comparison of mode of administration and an indirect test of convergent validity. While these different modes have their pros and cons, the key survey results are remarkably consistent across modes.

The survey was designed to meet or exceed the stringent protocols for stated preference surveys developed by the NOAA Panel on Contingent Valuation (Arrow et al. 1993) and the OMB (2003). One of these protocols stresses that the "commodity" being valued map closely to the underlying science. Following this guideline, we interviewed experts on ecological damages and important literature on the topic to develop a summary of the science (Cook et al, 2002).[i] The report serves as the foundation for the description of the park's condition and the commodity being valued, that is, the type and magnitude of improvements reasonably expected following further reductions of acid precursors. A major effort of our research was to accurately but meaningfully distill scientific information and convey it to a general audience. To this end, we convened 31 focus groups and conducted two major pretests to develop and assess alternative text, debriefing questions and graphics.

Guided by these groups and our review of the science, we consequently developed two survey versions, each with a particular scenario describing what will happen to the park's ecosystem absent a new policy intervention (the status quo), and how the ecosystem will improve with a proposed policy. These scenarios were intended to span the range of opinion about the future status of the park as of 2004. This design choice has the added advantage of permitting an external scope test of preferences, a key test of contingent valuation performance highlighted by the NOAA Panel. A common criticism of contingent valuation is that the hypothetical nature of the exercise tends to yield overestimates of WTP. In response, we typically followed a cautious or conservative approach in the survey design by characterizing the science, presenting information, and applying statistical methods in ways that are expected to yield estimates that understate rather than overstate the true WTP for the improvements described. For example, we do not characterize the policy intervention as "recovery" of acidified lakes. Rather, we characterize "improvement" in lakes that are affected by acidification. Also, we adopt the scenario with less extensive ecological improvement as our preferred scenario.

## III. SUMMARIZING OUR STATE OF THE SCIENCE REPORT

Significant reductions in sulfur dioxide and nitrogen oxide emissions resulting from the 1990 Clean Air Act Amendments (CAAA) have led to some recovery of acid-neutralizing capacity (ANC) and surface water pH in the Adirondacks, but not in proportion to the drop in emissions (Driscoll et al., 2001a; 2001b; 2003; Stoddard et al., 1999). Moreover, the future of the park's ecosystem depends importantly on whether, or when, the soils of the park's watersheds reach nitrogen saturation, which varies considerably by watershed. This variability and underlying uncertainty implies a range of possible future conditions of chronically acidic lakes from great degradation to a modest improvement.

The term "chronically acidic lakes" is in part a term of art. In the literature, one finds a variety of ANC levels used to define this, from ANC < 0 to ANC < 40 or 50. ANC < 0 cannot support fish life. Lakes with higher ANC, say up to 50 (but again there is no hard and fast rule here), can support fish, but they are generally considered impaired or sensitive. Sometimes, they are termed episodically acidic lakes, because much of the year they have acceptable ANC levels, which drop during snowmelt or rainfall and disrupt the lake ecosystem. That is, at high flow times during the year ANC could dip below 0 and be unable to buffer acidic inputs.

Research indicates that acidification also has harmed forests (Driscoll et al., 2001a; 2001b; Lawrence, 2001). In particular, acid deposition has been implicated in declines of high-elevation spruce stands. Moreover, there is mounting but as of yet not definitive evidence that damage to sugar maple and white ash stands also can be caused or exacerbated by acidification. There is also evidence that acidification is affecting some bird populations, especially water birds such as the common loon and hooded merganser. In addition, to the extent that forests are affected by acidification, songbirds nesting in affected trees may be affected as well.

Finally, our scientific review indicated that there remains uncertainty about the future status of the park in the absence of intervention and about the potential benefits of intervention. Nonetheless, focus group results clearly indicated that credibility of the survey depended on respondents believing that scientists have certainty in their understanding of the problem and how to fix it. If uncertainty in future status of the park with and without an intervention is described, respondents would reject the scenario.[ii] Thus, all scientific elements of the survey were treated as if they were certain.

## IV. ANALYSIS OF APPLICABILITY OF WTP ESTIMATES TO SECOND PROSPECTIVE STUDY

### A. From Literature Available to us at the time of the survey

<u>The Details of the Report mapped to the Survey</u>

**Baseline: Current.** As stated in our report, drawing on the Adirondack Lake Survey Corporation (ALSC) surveyed 1,469 lakes in the Adirondacks between 1984 and 1987, about 40% of the lakes in the Adirondacks are vulnerable to acid deposition and amenable to improvement by reducing it, with 10% being acidic from natural origins and not amenable to improvement. Of the 2,796 lakes with an area greater than 0.25 hectares within the Adirondack Park[iii], most are found at elevations below 2,100 ft while some very small lakes occur at elevations up to 3,900 ft (Driscoll et al, 1991). Although this survey is old, it was very comprehensive, in contrast to MAGIC modeling, where underlying runs may be based on a sample of as few as 44 lakes, which are then scaled to be representative of only a subset of all of the lakes. In this field survey, lakes were selected to provide a nonbiased, representative sample of lakes in the region based on lake size and elevation. Of the lakes surveyed, 24% had summer pH values below 5.0; in addition, 48% of lakes surveyed had ANC less than forty microequivalents per liter (<40μeq/L). In a related paper, Dr. Charles Driscoll (Syracuse University), stated that 27% of all lakes were chronically acidic (ANC<0) and another 21% were episodically acidic (0<ANC<40) (Driscoll et al. 1998). In another paper Driscoll et al (2001b) find that 41% of lakes in the Adirondacks exhibit chronic and/or episodic acidification. 83% of these impacted lakes are acidic due to atmospheric deposition. However, the U.S. EPA's Eastern Lake Survey (which underlies EPA (1995)) indicated much lower percentages of acidic and low-ANC lakes than did the ALSC survey. This was, in part[iv], because the EPA survey only included lakes larger than 4 ha, whereas the ALSC survey included lakes as small as 1 ha or less.

**Baseline: Future**. From our Summary of the Science Report, we stated that the uncertainty in estimating the time required for watersheds to reach nitrogen saturation implies a range for the future baseline of chronically acidic lakes (assuming constant future deposition). Such estimates range from a huge degradation to a modest improvement. Specifically, the percentage of lakes that are chronically acidic--19% in 1984--could rise to 43% or more by 2040 with saturation at 50 years and could fall to 11% or less by 2040 if we assume saturation is never reached.

Because of the scientific uncertainty in the estimated times to nitrogen saturation, EPA explored the impact of various assumed estimates on model forecasts of recovery rates in its 1995 Nitrogen Bounding Study (NBS). The reference point for the study was the 1984 National Surface Water Survey (NSWS). Nineteen percent of Adirondack lakes included in the NBS were chronically acidic (ANC<0 μeq/L) and 55% were "sensitive" (ANC<50 μeq/L). The NBS study projected the number of "acidic" and "sensitive" lakes in 2040 as a consequence of several different deposition scenarios. All scenarios assumed that the 1990 Clean Air Act Amendments would be fully implemented by 2010, which includes roughly a ten million ton reduction in $SO_2$ emissions and a two million ton reduction in $NO_X$ emissions from electric power plants, compared to previous trends. The maximum emissions modeled were a continuation of 2010 levels to 2040, and the minimum emissions modeled were a decrease to "background levels" (only airborne natural, agricultural fertilizer, and domestic livestock sources) in 2040.

The results of the NBS for chronically "acidic" lakes in the Adirondacks is[v] that assuming the maximum emissions (full CAAA to 2040), the percent of acidic study lakes could rise from 19% to 43% if nitrogen saturation is reached in 50 years. If, however, nitrogen saturation is never reached, the percent of acidic lakes would fall from 19% to 11% with 1990 CAAA levels. Note that if emissions were less than 1990 CAAA levels (additional reductions), the percent of acidic lakes would be smaller for all of the nitrogen saturation scenarios. The results were very similar for the projection of "sensitive" lakes. The 55% of lakes identified as sensitive in 1984 would increase to 67% if nitrogen saturation is reached in fifty years. If saturation is never reached, 54% of lakes were estimated to be sensitive in 2040 (again, assuming maximum emissions).

Our expert on the project, Dr. Driscoll, indicated that his expectations were, absent no further emission reductions, that these lake conditions would remain in the future or worsen slightly (Driscoll, pers. Comm., 1998).

**Setting the Ecological Baseline**. Using the above information, we assumed that with fully implemented CAAA (Title IV), a maximum of 40% of Adirondack lakes would be improved by decreases in deposition. These lakes were termed "lakes of concern" and are generally assumed to be lakes with ANC < 50. As seen above, this is a rough estimate based on one set of assumptions, however, and other estimates with different sets of assumptions could range from 32-50% (see endnote).[vi]

To bracket the range of uncertainty in the science and to create an opportunity to test the sensitivity of willingness to pay to the scale of ecological improvements, we employ a *base* and a *scope* version of the survey. The *base* version of the survey indicates that in 2003 about half of the lakes (1500 out of 3000) are injured due to acid deposition.

The *base* survey also indicates that there are minor injuries to high-elevation stands of red spruce, covering about 3 percent of the park's area, and injuries to two aquatic bird species, the common loon and hooded merganser. While these damages are considered more speculative in the science literature, we found that unless terrestrial damages were specifically acknowledged, focus group respondents would embed such damages (as well as reductions in such damages) when characterizing their willingness to pay.

**Intervention to Change the Baseline.** It was difficult to get scientists to provide estimates of the timing and extent of improvements to "lakes of concern" from reduced acid deposition because of the complex ecological and geochemical interactions. A set of "ballpark" recovery estimates was offered by Kretser (pers. Comm., Oct 22, 1998), assuming deposition ended today, is shown in Table 3.

**TABLE 1. ONE SCIENTIST'S OPINION OF THE TIME TO "RECOVERY" (DEFINED AS A RETURN TO A CHEMICAL STATE THAT WOULD AGAIN SUPPORT CERTAIN FISH SPECIES) FOR ADIRONDACK LAKES (OCT. 22, 1998 INTERVIEW WITH WALT KRETSER AS DESCRIBED IN COOK ET AL. 2002.)**

| TIME TO RECOVERY IF DEPOSITION ENDED TODAY | % OF LAKES | CHARACTERIZATION OF CURRENT ANC OR PH LEVELS (ANC IN μEQ/L) |
|---|---|---|
| N/A | 52 | ANC > 40 (not at risk) |
| 0-10 years | 20 | 40 > ANC > 0 |
| 10 to 20 years | 3 | ANC <= 0 and/or pH <= 5.0 |
| 50 to 100 years | 10 | ANC <= 0 and/or pH <= 5.0 |
| geologic time (centuries) | 10 | ANC <= 0 and/or pH <= 5.0 |
| never, naturally acidic | 7 | ANC <= 0 and/or pH <= 5.0 |

Other researchers, including Driscoll and Simonin (pers. Comm., Oct 23, 1998) were a bit more optimistic when posed with the same scenario. Driscoll (pers. Comm., Oct 23, 1998.) thought that if sulfur and nitrogen deposition were eliminated, pH and ANC would still not *immediately* increase. It will take from years to decades for all of the exogenous sulfur to be removed from the system (although, as noted, the 1990 CAAA have achieved observable improvements in sulfate concentrations), and probably much longer for the excess nitrogen to be removed. Furthermore, biological recovery (forests, fish and wildlife populations) will most likely take longer than chemical recovery of streams, lakes and ponds.

The 20% of lakes experiencing recovery within 10 years, and 33% of lakes recovering within 100 years suggested by Kretzer was also justified in part from the analyses described in EPA (1995), which found that under dramatic reductions in emissions (to background) 18% of the lakes would no longer be chronically acidic as of 2040. For comparison, EPA (1995) also modeled a Title IV to beyond-Title IV scenario, finding, with reasonable assumptions, only a 4% improvement in chronically acidic lakes by 2040, and a 2% net improvement in episodically acidic lakes (from a total of 43% lakes of concern with Title IV to 41% lakes of concern beyond Title IV). This scenario involved a 44% reduction of SO2 emissions from all sources beyond Title IV and a 24% reduction of NOx emissions. This closely matches the CAIR-plus scenario in the Second Prospective Study, which features emission reductions of 47% and 28% for SO2 and NOx, respectively.

**Setting the Survey Scenarios.** Both the *base* and the *scope* surveys propose an intervention program that would involve improving the health of the lakes by liming them. Thus, there was no explicit link of our survey to changes in emissions. The *base* survey said that the program would improve about 600 lakes (20% of 3,000 lakes), with only very minor improvements to birds and forests, based on Kretzer and EPA (1995), above. Wording in the survey is carefully constructed to avoid characterization of restoration, in part because the ecological profile of a recovered lake does not necessarily mirror its previous patterns, and in part because of uncertainty about the time to observing chemical and biological responses from changes in emissions and deposition. Hence, we used a cautious approach to valuing benefits required that described “improvement” in the lakes of concern instead of “restoration.”

Note, that both sources use an ending or near ending of deposition to motivate their estimates, although extending the time to recovery beyond 10 years adds another 3% and to 50-100 years adds another 10% of lakes that could be “recovered.” giving us some justification for assuming that a policy reducing emissions less than required to reach background could be expected to result in a recovery at 20% of the lakes of concern. In our study, the *base* survey indicates that improvements would be realized by 2014, however, these does not literally correspond to a reasonable timeframe to recovery. We felt an accelerated characterization of the timing of improvements in lake status was justified because we found that time horizons as distant as 2040 were not viewed as credible by focus group participants.

The *scope* version of the survey also indicates that about half the lakes are currently injured and, absent intervention, an additional 5% of the lakes would worsen over time. While it suggests worsening future conditions, it also indicates that by 2014 all but 10% of the lakes would be improved by the liming program. The *scope* version also reports broader terrestrial ecosystem damages than the base survey. Specifically, this version describes injuries to sugar maple and white ash stands, as well as red spruce, covering 10 percent of the park's area. It also describes injuries to songbirds in these affected forests, namely the wood thrush and tree swallow, as well as the aquatic birds. Absent intervention, the health of these species would gradually worsen. The intervention program would also involve liming the forests and would improve them to a 99% healthy state.

In what follows, we focus entirely on the characterization of ecosystem damages and improvements in the base survey, as the scope survey describes improvements that are simply too optimistic from any reasonable emissions control scenario in light of current scientific understanding. In the following sections we review previous analyses of the likely effect of changes in acid deposition on the health of Adirondack lakes.

<u>Implications for using WTP Values from Adirondacks Survey</u>

Assuming the state of the science is unchanged from our Summary of the Science Report, we suggest that our WTP estimates from the base version of our survey can best be thought of as an upper bound to apply to the lake improvements related to the CAIR-plus scenario versus a Title IV baseline. We believe that our baseline is consistent with the literature summarized in our state of the science report. However, the improvement may be on the high side. In particular, we are concerned that a scenario and base case modeled in EPA (1995) that reasonably matches that for Title IV to CAIR-plus estimates only a 2-4% recovery in lakes of concern, depending on how lakes of concerned are defined.

At the same time we have less uncertainty about our base case results applying to the improvement in lakes that encompass the entire period from pre-CAAA to the CAIR rule and other regulations. We base this on the logical point that the improvements in lakes from pre-CAAA to post Title IV are larger and may be better represented by a 20% improvement.

While the WTP estimates for the base version of our survey may be appropriate for the Second Prospective Study, the scope version results are not. The scope version is more optimistic about the degree of improvement than is possible. Thus, we do not recommend its use.

It would be highly unlikely that the RFF scenarios were matched perfectly to the MAGIC estimates of lake status from the Second Prospective Study. In the event that a more persuasive case for the use of our estimates was desired, we offer a recommendation

1. We would be willing to mount an additional survey (either by mail or through the KN internet panel) sufficient to examine the effect on WTP of any change in the commodity required to match the estimates provided by MAGIC. The scope of this survey may be large or small. We recommend a conjoint survey so that WTP could be estimated for any changes in lake status associated with different emissions (deposition) scenarios modeled with MAGIC as part of the Second Prospective Study. The scope of this work would not be as extensive as initiating a whole new survey, given that key components of the survey have already been tested and that we are already engaged in similar work as part of our ongoing study of the benefits of reduced acidification in the Southern Appalachians.

There is a major caveat to all the preceding discussion, however. As noted above, we shortened the period before improvement in the lakes would be realized following emission changes associated with any of the scenarios being discussed. This leads to an overestimate of WTP if the individual rate of time preference is positive (Note that people still would pay beginning today). It is a simple matter to adjust our WTP estimates given an acceptable assumption of a rate of time preference.

Additional Issues In Using Our Study Results

There are two additional issues with using our study results. The first concerns the spatial extent of the market. Our survey was administered to an RDD-recruited internet panel limited to adults living in New York State, as well as through a mail survey of an RDD-matched sample of New York State adults. Given the results of a variety of tests of sample representativeness, we feel that the sample and WTP estimates are reasonably representative of New York State households. Whether these results can be applied outside of New York State, however, is another question. We found that WTP declined with distance from residence to the Park. This relationship could be used to extrapolate beyond New York State. However, in our judgment, this would be risky as the survey mechanism for eliciting WTP was an increase in state taxes that would go into an Adirondacks fund. It is possible that residents of other states would not see the Adirondacks as part of their state's responsibilities and that therefore, there would be a significant discontinuity of the WTP gradient at the New York State border. This proposition could be tested by mounting a new survey in other states, but the survey would not have to be a WTP survey and could be very simple.

The second issue relates to the intention of the Second Prospective Study team to compare use values for New Yorkers to our total value estimates. We found that frequent visitors to the Adirondacks tend to have a larger WTP than non-users, when "use" was defined as fairly intense use (over 10 visits per year). However, we did not estimate a separate use value. Also, our "commodity" was a 20% improvement in lakes of concern. We doubt that the RUM analysis will be easily comparable. Thus, caution will be called for in comparing any use value estimate to our total value estimates.

### B. Additional Literature Assessment

There are three additional studies relevant to an assessment of whether our Summary of the Science holds. The first is an IEc report, *Economic Benefits Assessment of Decreased Acidification of Fresh Water Lakes and Streams in the United State Attributable to the 1990 Clean Air Act Amendments, 1990-2010* (IEc, 1999). The second is NAPAP (2005) and the third is the memo describing emissions scenarios for the Second Prospective Study.

<u>IEc Report</u>

IEc conducted a study using the MAGIC model that was cited in the First Prospective 812 Study (IEc 1999 as provide in USEPA, 1999, Appendix E). The relevant MAGIC analyses were based on a sample of 33 Adirondack lakes with an ANC < 400. The model was used to estimate the health of the Adirondack lakes with and without Title IV in 2010 (rather than 2040). While estimates of percentage improvement (extrapolated to a larger sample of Adirondacks lakes) were provided for pH categories rather than ANC categories, the pH categories correspond approximately to ANC < 0 and ANC < 5 (inclusive of ANC < 0). These MAGIC analyses assumed nitrogen saturation by 2010. The most relevant statistic is that in 2010 17% of lakes were projected to have ANC <5 with Title IV and 5% (of the 17%) would have ANC < 0. The results of this analysis are reported in Table 2.

At first blush, this estimate seems very optimistic relative to the lake conditions expected in the EPA (1995) study (13% of lakes with ANC < 0) or the estimate provided by Dr. Driscoll (25%). However, with lake ANC status generally expected to be stable or worsening between 2010 and 2040, these estimates could be viewed as underestimates for 2040. Furthermore, the IEc estimate does not account for episodically acidic lakes between ANC 5 and 50. Also, the studies appear to have different definitions of the relevant population of Adirondacks lakes.

IEc (1999) also employed MAGIC to estimate lake health assuming a without CAAA (or "pre-CAAA") counterfactual case. These results are also reported in Table 2. They show a baseline of about 22 percent of lakes with ANC < 5, which is not inconsistent with the base survey's characterization that half of the lakes would be acidic absent a program.

Table 2 can also be used to examine the improvement in lakes as a result of Title IV. Irrespective of the assumption about nitrogen saturation, the projections yield a 5% reduction in ANC < 5 lakes by 2010. This small improvement is far less than the 20% improvement described in the survey, but again, improvements to episodic lakes are not accounted for and the time allowed for improvements is much shorter than that used by Kretzer (1998) or EPA (1995).

**TABLE 2. PROJECTED 2010 LAKE ANC STATUS WITH AND WITHOUT NITROGEN SATURATION IN IEC (1999)**

| ANC CLASSIFICATION | % LAKES (WITHOUT N SATURATION) | # OF LAKES | % LAKES (WITH N SATURATION) | # OF LAKES |
|---|---|---|---|---|
| WITH CAAA (Title IV) <0 µeq/L | 2 | 56 | 5 | 140 |
| <5 µeq/L | 18 | 504 | 17 | 476 |
| WITHOUT CAAA <0 µeq/L | 6 | 168 | 6 | 168 |
| <5 µeq/L | 23 | 644 | 22 | 616 |
| Total Lakes | | 2800 | | 2800 |

Note that this table has an odd finding. We would expect that lake status would be worse in a world with N saturation than without it. Without N saturation, nitrogen deposition would get absorbed in the ecosystem with less ending up in lakes. Yet, table 2 shows some results where lakes are better off with than without N saturation.

NAPAP 2005

NAPAP (2005) also presents analyses using MAGIC. One scenario assumes full implementation of Title IV while another assumes a temporal emissions profile roughly equivalent to that expected after CAIR rule. The report shows that absent Title IV, the percent of chronically acidic lakes would be 40% in 2030, up from 33% in 1984. With Title IV, 12% of lakes in the Adirondacks would be chronically acidic (ANC < 0) in 2030. The 12% estimate is larger than that from IEc above, although the simulation years are different (2030 vs. 2010).

More striking is the improvement from Title IV implied by the NAPAP results -- 28% of ANC < 0 lakes (i.e., 40% - 12%), even larger than our 20% estimate for ANC < 50 lakes! Note also that no change in the number of non-acidic lakes is expected. The report states that 36% of the lakes will be non-acidic in 2030 with Title IV, unchanged from 1984. If non-acidic lakes are defined as ANC 50 or greater, this finding implies that Title IV *had no effect* on the category ANC < 50 lakes, even though it had a large effect in moving chronically acidic lakes to the episodic category.

The finding that 36% of the lakes will be non-acidic in 2030 also implies that 64% of the lakes will be acidic in 2030 with Title IV, which is more pessimistic than the 50% reported in our base survey as well as the estimates reported in EPA (1995).

Turning to change from Title IV to CAIR, NAPAP (2005) predicts that the 12% of lakes that are chronically acidic will have non-negative ANC values by 2030 with reductions in emissions associated with CAIR. To see whether this is consistent with our 20% improvement, we need to know the status of episodic lakes with CAIR. The report is silent on this exact issue, but says that there will be no improvement in such lakes for a scenario similar to CAIR. However, for a "beyond CAIR" scenario, which arguably could match with the Second Prospective Study (because of the many post-Title IV regulations being considered), the percentage of lakes that are episodically acidic did decrease.

At this point it is worth noting that in a very recent conversation with Jack Cosby, we asked him about the finding in these MAGIC runs and in others that episodic lakes (0<ANC<50) do not appear to improve, in the sense of moving above the ANC=50 threshold. We were told that these runs are often performed on a sample as small as 44 lakes that is then scaled to only a subset of Adirondacks lakes in a separate step. It is possible that none of these particular lakes has a baseline ANC close enough to 50 to be sent over the threshold by CAIR emissions reductions. That is, there could be a sampling bias.

Emissions Assumptions from the Second Prospective Study and Other Reports

In our study, our intention was to compare a scenario assuming full implementation of Title IV with a reasonable emission control scenario beyond Title IV. This is contrary to the intention of the Second Prospective Study, which is to compare a scenario with the CAAA (that now goes quite a way beyond Title IV (e.g., CAIR, Diesel Rule, etc.) to a scenario absent the 1990 CAAA. Nevertheless, our results could conceivably comport most with the latter scenario.

One way to make such judgments is to compare the emissions baselines and changes underlying the various scenarios in EPA (1995) and the First and Second Prospective Studies (see Table 3). It is not straightforward to compare emission profiles in these documents because the end dates are different (2010 ($1^{st}$), 2020 ($2^{nd}$) and 2040 (EPA, 1995)). Furthermore, these estimates mask spatial differences in emissions and thus deposition that may be meaningful.

We first compare $SO_2$ emissions across the different studies. Without the CAAA, the emissions estimates in 2010 in the two Prospective Studies and the 1995 EPA report are quite similar (ranging from 18 -19 million tons from utilities, and are around 23 million tons from utility and industrial emissions). Post-CAAA, SO2 emissions are higher in the First Prospective Study (9.8 million tons) than in EPA (1995) (9.5 million tons). This is in spite of the fact that the First Prospective Study's definition of the 1990 CAAA includes programs that the EPA (1995) report did not.[vii] Thus assumptions about demand growth, etc. may explain the difference between these estimates. Emissions in the Second Prospective Study (which includes CAIR and a host of further regulatory changes) naturally exceed those of the First Prospective Study (9.8 million tons versus 6.3 million tons). For our purposes, the most relevant comparison is between the EPA (1995) report and the Second Prospective Study given implementation of Title IV. $SO_2$ emissions for the latter are only about 60% of the former, while NOx emissions for the latter are only 40% of the former.

In terms of $SO_2$ emissions changes, the estimated emissions reductions resulting from the adoption of the 1990 CAAA in the Second Prospective Study are about 13 million tons, which is about the same quantity of emissions reductions EPA (1995) estimated would occur if emissions were reduced from Title IV levels to background levels. Thus the 20% of lakes improving reduction scenario in our base survey, to the extent it was based on EPA (1995), may be applicable to the entire change in $SO_2$ emissions from pre-CAAA to CAIR-plus.

Of course, it is insufficient to just look at $SO_2$ changes. We also need to compare NOx emissions between the two reports. Comparable to the proceeding point, the EPA (1995) NOx reduction from Title IV levels to background levels is 9 million tons, while the

Second Prospective Study's CAIR-plus reduction relative to pre-CAAA is about 8 million tons. This reinforces the idea that the EPA (1995) runs are broadly applicable to the entire change in deposition from pre-CAAA to CAIR-plus.

### TABLE 3. EMISSIONS ASSUMPTIONS BY STUDY

| SCENARIO | SO2 EMISSIONS IN 2010 | NOX EMISSIONS IN 2010 | N SAT. BY END YEAR? | END YEAR |
|---|---|---|---|---|
| CAAA (1995 EPA) | 9,519,000 tons utility (or 13,714,000 with utility + industry) | 10,700,000 tons (utility + industrial) | no | 2040 |
| w/o CAAA (1995 EPA) | 18,685,000 tons utility | 11,819,456 tons (utility + industrial) | | 2040 |
| Background (NAPAP 1990) | 460,000 tons | 2,000,000 tons (lightning + soil) | no | 2040 |
| Background (1999 EPA) | 0 | 0 | | |
| CAAA w/ Title IV (1999 IEc) | 9,860,800 tons utility (or 15,854,700 with utility + industrial) | 5,950,800 tons (utility + industrial) | no | 2010 |
| w/o CAAA (1999 IEc) | 17,696,000 tons utility (or 23,689,900 tons utility + industrial) | 12,632,600 tons (utility + industrial) | ? | ? |
| CAAA (2006 EPA, Second Prospectus) w/ CAIR, etc. | 6,365,458 tons utility (or 8,532,848 tons utility + industrial) | 4,283,120 tons (utility + industrial) | ? | ? |
| w/o CAAA (2006 EPA, Second Prospectus) | 18,867,532 tons utility (or 23,628,787 tons utility + industrial) | 11,904,202 tons (utility + industrial) | ? | ? |

## V. CONCLUSIONS AND RECOMMENDATIONS WITH RESPECT TO NEW INFORMATION

Without access to the MAGIC analyses for the Second Prospective Study, it is impossible to be definitive about the relationship between our estimates of WTP and the baseline and lake improvements associated with that study. Therefore, our main recommendation is:

1. Provide RFF with MAGIC runs, along with their assumptions about the relevant population of lakes, when they become available and we will analyze them to determine the suitability of out WTP estimates.

Based on the reports described above, a picture emerges that is different than described in our Summary of the Science report and surveys. The MAGIC runs in NAPAP (2005) and IEc (1999) (as well as other runs not detailed above) indicate lake conditions post-Title IV that are healthier yet predict lake improvements that are smaller than is described in our base survey. Indeed, if lakes of concern are defined as chronically and episodically acidic lakes with ANC < 50, these recent runs show *no improvements*. Rather, all remaining chronically acidic lakes that can be improved are improved to above ANC =0, but no episodically acidic lakes cross over the ANC = 50 boundary. On the basis of comments from Jack Cosby (personal communications, September 26, 2006) we may question this result on the sample of lakes study is different than the population we refer to in the survey. Nevertheless, if these results are taken at face value, our WTP estimates from the base version of our survey clearly overestimate the WTP for the improvements expected to occur in the Adirondacks as the result of reduced deposition associated with a CAIR-plus regulatory scenario versus Title IV, as laid out in the Second Prospective Study plan.

However, our base case results may be appropriate for valuing the improvement in lakes that encompass the entire period from pre-CAAA to the CAIR rule and other regulations. We base this on the very close correspondence between emissions reductions over this period as described in the Second Prospective Study Plan and the reductions modeled in EPA (1995), as well as on the logical point that the improvements in chronic lakes in NAPAP 2005 ascribed to CAIR over Title IV could only be increased if the pre-CAAA to Title IV emissions reductions were included. Further, the finding that chronically acidic lakes were reduced 28% from 2030 levels as a result of Title IV further emboldens us to conclude that our WTP estimates are not overestimates and may, indeed, be underestimates with this alternative emissions scenario comparison.

Our confidence in these conclusions would be increased, however, if we examined the IEc study underlying the First Prospective Study, reported on in Appendix E, and the MAGIC runs underlying NAPAP 2005. Also, there are a variety of more recent MAGIC runs and reports about these runs that could also be analyzed. Thus, another recommendation is to:

2. Analyze MAGIC runs and studies by IEc (1999), NAPAP (2005) and others as appropriate to see if the above conclusion should be modified.

While the WTP estimates for the base version of our survey may be appropriate for the Title IV to CAIR-plus scenarios, the scope version results are not. The scope version is much more optimistic about the degree of improvement than is possible (partly because the baseline we assumed may have been too pessimistic!). Thus, we do not recommend its use here.

However, on the basis of NAPAP 2005, one could argue that our *scope* case estimates actually are a fair match for the entire change in lake quality from pre-CAAA to CAIR-plus. After all, the scope case presents a slightly worsening baseline, as in NAPAP without Title IV. And NAPAP's improvement over the entire period is from 40% to 12% to 0% chronically acidic lakes, i.e., a 40% improvement, which is exactly what we provided in the scope case improvement survey (although our "lakes of concern" were intended to represent both chronically and episodically acidic lakes in the way we described their condition on the survey).

As noted above, because it would be highly unlikely that the RFF scenarios were matched perfectly to the MAGIC estimates of lake status from the Second Prospective Study:

3. RFF would be willing to mount an additional survey (either by mail or through the KN internet panel) sufficient to examine the effect on WTP of any change in the commodity required to match the estimates provided by MAGIC.

And, as above, there is a major caveat to all the preceding discussion, however. Our WTP estimates need to be adjusted for the shortened period of improvement, given an acceptable assumption of a rate of time preference.

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## ENDNOTES

[i] A draft was peer-reviewed by advocates and by scientists at the NY Department of Environmental Conservation.

[ii] This is labeled "ambiguity aversion" in experimental studies (Van Dijk and Zeelenberg, 2003).

[iii.] Technically, there are 2,796 lakes within the Adirondack's ecological zone, which is defined as the area lying within the 1,000 ft elevation contour. It roughly corresponds to the political boundary of the Park.

[iv.] It should also be noted that the ELS survey in the Adirondack region involved only 153 waters and that this small a sample may have also influenced the percentages reported.

[v.] It should be noted, however, that the NBS was performed for several East Coast watersheds, and thus the range of times to nitrogen saturation does not necessarily reflect the range of scientific opinion specific to the Adirondacks, but rather general design values chosen by EPA. One researcher felt that a scenario including a time to saturation of 25 years should have been included (Simonin, pers. comm.).

[vi.] This number is derived by simply adding the percentage of thin-till lakes and percentage of mounded seepage lakes. It assumes the following:

- Thin till and mounded seepage lakes are those most susceptible
- The percentage of thin till and mounded seepage lakes in the ALSC sample is representative of their percentage in all Adirondack lakes.

There is uncertainty about the 40% number. Currently this number includes all mounded seepage and thin till drainage lakes, both low and high DOC (3+3+19+15 in the Table). The logic is that the geology of these types of lakes makes them vulnerable, and even high DOC lakes should be included because they may have been acidified beyond their natural state. An alternative reasoning would yield 32%, which is the percentage of low-DOC lakes for the first four classes in the table. This ignores lakes with some degree of natural acidity (high DOC). Finally, another alternative would be to use the actual percentage of sensitive (ANC<100 $\mu$eq/L) lakes (around 45-50%).

[vii] For example, the First Prospectus included a program similar to the $NO_X$ SIP Call referred to as an OTAG $NO_X$ reduction program.