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MEMORANDUM

To: Willard Smith
From: Kathleen Bell, Leland Deck, Greg Michaels, and Chris Paciorek
Date: January 17, 1994
Re: Inventory of Ecosystem Effects

The purpose of this memorandum is to provide an inventory of ecosystem benefits categories germane to reductions in nitrogen oxide emissions. This corresponds to Task 2 (Literature Search and Definition of Benefits Categories) of SAIC Work Assignment 1-47. The inventory is meant to encourage a broadening of the scope of benefits estimated for nitrogen oxide emissions reductions. The inventory is designed to facilitate the valuation of 'ecosystem' benefits.

The valuation of ecosystem benefits associated with reductions in NO, emissions involves establishing four sets of linkages:

(1) predicting the changes in emissions and determining the associated air quality distribution;

(2) assessing the physical effects of these changes in emissions and air quality on ecosystems;

(3) evaluating the resulting changes in ecosystem service flows from these physical effects; and

(4) assigning monetary values to the changes in ecosystem services flows.

The first linkage is used to identify the extent and location. of emission changes from the proposed regulation and to isolate geographically relevant ecosystem categories. This linkage was modeled by SAI using the Urban Airshed Model (UAM-V) and the Regional Transport Model (Version III RTM). Building on the results of this modeling exercise, the remaining linkages involve characterizing the dynamics between nitrogen oxide emissions and ecosystems. This memorandum addresses these linkages, presenting the results of a literature search on the physical effects of nitrogen oxides on ecosystems as well as a limited discussion of the types of information required to quantify and monetize the physical effects on various services by ecosystem categories.

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1. Introduction

A. Regulatory Background

Under Title IV of the Clean Air Act Amendments (CAAA) of 1990, US EPA has promulgated standards for NO_x emissions. These standards are directed at selected power plants and are intended to assist state and local pollution control agencies with the meeting of ambient air quality standards. The proposed Phase I rules will affect approximately 179 coal-fired boilers at 110 locations. Seven compounds are typically designated as nitrogen oxides or NO_x: NO (nitric oxide), NO_2 (nitrogen dioxide), N_2O (nitrous oxide), NO_3 (nitrogen trioxide), N_2O_3 (dinitrogen),N_2O_4 (dinitrogen tertoxide), and N_2O_5 (dinitrogen pentoxide). Emissions from the stacks of power plants are typically in the form of nitric oxides (NO). These oxides react in the atmosphere to form NO, and other pollutants such as ozone. In addition, emissions of nitrogen oxides are often transformed into the acidic compounds termed nitrates. These acidic compounds often make their way into ecosystems in the form of wet or dry deposition. When assessing alternative control strategies, state and local pollution control agencies award much emphasis to benefits such as improved health and visibility or reduced materials damages.. Little attention is directed towards more nontraditional benefit categories such as ecosystem effects, and it is the purpose of this memorandum to offer insights concerning these categories and their likely magnitude and/or importance.

B. Ecosystem Effects

The effects of nitrogen compounds on ecosystems are difficult to assess because they are dependent on several complex processes including chemical, atmospheric, and biological. Valuation of ecosystem changes is a similarly complex task that encompasses modeling psychological and economic processes. Only recently has much research focused on whether or not atmospheric deposition of inorganic nitrogen has made an impact on aquatic, terrestrial, and other types of ecosystems. This research has been motivated by the growing atmospheric concentrations of nitrogen compounds and new discoveries about nitrogen saturated ecosystems. Analyses have focused on identifying potential acidification and nitrogen loading effects on ecosystems. For example, nitrogen deposition has been studied in relation to red spruce forest decline in the northeastern portion of the United States (NAPAP 1991a) and to eutrophication problems in the Chesapeake Bay Region (Fisher et al. 1988 and D'Elia 1982). This inventory summarizes the results of past and ongoing efforts to understand the dynamics of ecosystem impacts of nitrogen oxide emissions. In addition to devoting attention exclusively to nitrogen oxides, some effort is extended to assess the effects of nitrogen oxide controls on ozone levels and the impact of changes in ozone levels on ecosystem service flows. The photochemical production of ozone (O_3) is the result of atmospheric physical processes and chemical processes involving volatile organic compounds (VOC) and nitrogen oxides (NO_x). In areas where the ratio of VOC to NO_x emissions is high, nitrogen controls might result in reduced ozone levels. Ozone has been identified as damaging to forest and other terrestrial ecosystems.

Acidification effects are related to changes in the acidity of soil and water components of ecosystems. In the atmosphere, nitrogen oxides emissions are often chemically converted to acidic compounds known as nitrates. These nitrate compounds enter ecosystems in wet and dry forms including rain, snow, gases, fog, cloud droplets, and particles. Acidic deposition is associated with effects such as leaching and removal of nutrients from tree foliage, changes in soil and surface water chemistry and nutrient availability, and adverse biological effects in sensitive lakes, streams, forests, and other ecosystems (NAPAP,1992). The responses of terrestrial and aquatic ecosystems to nitrate depositions will vary according to the chemical composition of the receiving soils and waters.

Nitrogen loading effects are associated with the entry of all forms of nitrogen into ecosystems (i.e., forms other -than nitrates). The effects of any nutrient upon ecosystems depends on the amount of that nutrient in the system, the biological demand for that nutrient, and the rate and timing of deposition of the nutrient into the ecosystem. Nitrogen input levels that overstep the level of biological demand will result in negative growth responses or toxic effects. Alternatively, under other input and biological demand relationships, these inputs might have. a positive (deficiency situation) or neutral (sufficiency situation) effect (US EPA, 1991). For example, nitrogen loading effects will vary considerably in aquatic and forest systems, for the latter are commonly nitrogen limited while the former are not.

Ecosystem responses to ozone pollution include physical effects such as leaf and foliar injury as well as structural effects such as changes in species composition. For example, physical effects such as foliar injury and reduced photosynthesis, carbohydrate production and allocation, plant vigor, growth, and reproduction have been observed in eastern white pines in the Appalachian Mountains and major structural changes have been documented in the San Bernardino National Forest Ecosystem where sensitive species such as ponderosa and Jeffrey pine have been out-competed by other species (US EPA, 1993).

C. Ecosystem Categories and Associated Service Flows

As noted above, acidification, nitrogen loading, and ozone effects will vary according to the biological demand relationships and physical background conditions of ecosystems. Within this memorandum, attempts are made to broadly group types of ecosystems by background conditions and biological structure. However, there is likely to be considerable variation within ecosystem categories across the United States. To facilitate comparisons and inferences regarding ecosystem responses, ecosystem categories have been designated as follows: forests, freshwater wetlands, coastal wetlands, streams and lakes, coral reefs, oceans, deserts, soils, and grasslands. For each ecosystem category, several potential service flows were examined. Direct service flows referenced include hiking/hunting/camping, birdwatching and wildlife viewing, scenic beauty, boating/water activities, commercial fishing, recreational fishing, and waste sink services; while indirect service flows explored include storm protection/wave buffering, flood control, nutrient removal, pollutant uptake, sediment control, biodiversity preservation, and agricultural pest control.

As a first step, service provision by ecosystem category was broadly characterized. In cases where the ecosystem did provide a service, the provision was designated as dependent on biotic, abiotic, or both abiotic and biotic components of the ecosystem. Abiotic properties might include climatic conditions or soil or sediment types, whereas biotic properties include ecosystem structure (i.e., species) and ecosystems functions (i.e., ecosystem energy source and nutrient processing). **Exhibit 1 [Inventory of Ecosystem Services]** presents a matrix of services by ecosystem category. Cells in the matrix that are shaded gray indicate provision, while white cells indicate no provision. The letter designations of 'A' and 'B' . represent dependence on abiotic and biotic components of the. ecosystem. For example, the biotic components of forests provide numerous recreational opportunities such as hiking, hunting, camping, birdwatching, and wildlife viewing as well as indirect services such as flood control, nutrient removal, pollutant uptake, and sediment control.

The recreation-based direct service flows (i.e., hiking, hunting, boating, and fishing) provided by ecosystems are quite tangible and in turn well understood by both individuals and welfare economists. Demand for such service flows have been well studied as have the linkages between various policy changes or events and the service flow demands. Walsh et al. (1992) provide a thoughtful discussion of recreation demand studies completed over the period from 1968 to 1988. Many of the indirect or non-use service flows are less well-studied and recognized by individuals. Waste sink services refer to the capacity of streams and lakes as well as soils to hold and process certain waste streams. Wetlands and coral reefs provide storm protection and wave buffering services by maintaining water in their ecosystems. Similarly, flood control service flows are provided by forests, wetlands, and streams and lakes. These ecosystems store floodwaters and gradually release these waters through nondisturbing processes. Nutrient removal services refer to the ability of forests and wetland ecosystems to convert deposited nutrients into forms used by plants and other organisms. The removal of nutrients typically results in improvements of water quality which in turn increases the quality of recreational service flows and drinking water supplies. Analogous dynamics characterize the service flows termed pollutant uptake where ecosystems absorb or remove pollutants from waters. Sediment control service flows involve the stabilization of sediment flows carried by waters through forest, wetland, and streams and lake ecosystems. Stabilization involves the trapping of sediments and erosion control. Biodiversity preservation service flows refer to the habitat quality of various ecosystems to support numerous species and genetic diversity.

After designating the various service flows provided by different ecosystems, it is then necessary to examine whether or not the biotic components of ecosystems involved in the provision of such flows are susceptible to air pollution, particularly nitrogen oxides or ozone levels. The second step of this analysis involves establishing connections between the provision of a service and nitrogen oxide levels. The identification of these connections relies heavily on the scientific literature describing the physical effects of nitrogen oxides on various ecosystem components. The level of certainty attributed to these connections varies considerably across ecosystem,categories. The complexity of the nitrogen cycle makes for difficult assessments and has allowed for considerable debate within the scientific community regarding the nature and extent of nitrogen oxide effects on ecosystems.

Exhibit 1
Inventory of Ecosystem Services

End Points or Services	**Forests**	Freshwater Wetlands	Coastal Wetlands	Streams & Lakes	Coral Reefs	Oceans	Deserts	Soils	Grasslands
Direct Services									
Hiking/Hunting /Camping	B	B	B				B		
Birdwatching/ Wildlife Viewing	B	B	B	B		B			
Scenic Beauty	AB	AB	AB	A		A	A		A
Boating/Water Activities		A	A	A	AB	AB			
Commercial Fishing		B	B	B	B	B			
Recreational Fishing		AB	AB	AB	AB	AB			
Waste Sink				A			A	AB	
Indirect Services									
Storm Protection/ Wave Buffering		AB	AB		AB				
Flood Control	B	A		A					
Nutrient Removal									
Recreation	B	AB	AB						
Drinking Water	B	AB							
Pollutant Uptake									
	B	AB							
Recreation	B	AB							
Drinking Water	B	AB							
Industrial									
Sediment Control	B	AB	AB	AB					
Biodiversity Preservation	B	B	B	B	B	B			
Agricultural Pest Control	B	B	B						

Ecosystem Services have been ranked as follows: B (biotic), A (abiotic), and AB (both biotic and abiotic). Shaded boxes indicate provision of services.

Finally, the connections between physical effects and service flows are linked with economic studies of such or similar effects on ecosystem service flows to generate estimates of the monetary value associated with the prescribed changes in service flows.

The memorandum is divided into several ecosystem category-specific discussions. Within each discussion, a brief description of the ecosystem is offered, followed by an assessment of potential physical effects and the impacts of these effects on service flows. Results are presented in the following order: (2) forests, (3) freshwater wetlands, (4) coastal wetlands, (5) streams and lakes, (6) oceans and estuarine ecosystems, (7) deserts, and (8) grasslands. Coral reefs were dropped from the group of ecosystem categories referenced, as effects on these systems seemed extremely unlikely. The discussion of effects on soil ecosystems is incorporated within the discussion of other ecosystems such as forests, wetlands, lakes, and streams. Some sampling of economic studies was completed to characterize the breadth of analytical frameworks. This sampling does not represent a complete inventory for any particular ecosystem category.

The thought experiment offered by the proposed Title IV regulation involves delineating the impact of nitrogen oxide emission reductions. The nature of this policy defines the manner in which ecosystem effects are to be contemplated and raises a troublesome dichotomy between relevant economic and scientific research. Scientists tend to evaluate the impact of increases in emissions on ecosystems and therefore explain the causes of damage. In turn, economists concentrate on willingness to pay measures for reductions in emissions and therefore value the benefits of improvements. This subtlety is important in the context of this analysis, for if scientific linkages are to be adopted, it must be assumed that effects are reversible. In other words, if scientists can explain the effect of a given increase in emissions on an ecosystem, this information is more valuable when and if the effect of an equal reduction in emissions has a parallel effect in the opposite direction. The baseline emissions scenario for this analysis are current emissions levels. Nitrogen oxide emission levels have remained somewhat constant in recent years. From the period of 1983 to 1992, annual levels of NO, consistently neared 20 million metric tons, with some subtle evidence of an increasing trend over this time period (US EPA, 1994). It must then be assumed that ecosystem effects have already occurred and that the proposed reductions will reverse the negative impacts of such pollution.

2. Forests

A. Service Flows

As indicated in **Exhibit 1 [Inventory of Ecosystem Services],** forests provide many interconnecting service flows to individuals. For direct service flows, these range from recreational services such as hiking, hunting, camping, birdwatching, and wildlife viewing to nonuse services such as scenic beauty. For indirect service flows, products consist of flood control, nutrient removal, pollutant uptake, sediment control, biodiversity preservation, and agricultural pest control.

B. Physical Effects

Research on acidification and nitrogen loading effects as well as other air pollution related effects has been motivated by the recent decline of forest systems in central Europe and the United States. In the United States, forest decline has occurred in the red spruce on the East Coast, the commercial yellow pines in the Southeast, the New Jersey Pine Barrens, and in sugar maples in the Northeast (Mackenzie et al., 1989). The causes of these declines are largely uncertain. NAPAP organized a Forest Response Program (FRP) in 1985 to evaluate the significance of forest damage caused by acidic deposition, the causal relationships between air pollutants and forest damage, and the dynamics of these relationships on a regional or stand basis. Research has been divided across four regions: Eastern Spruce-Fir, Southern Commercial Forests, Eastern Hardwoods, and Western Conifers. Numerous models have been developed as part of this program including Branch/Foliage 3 Model (Development and Physiology of a Single Branch), the Simple Whole Tree Model (Mature Tree); the ROPIS-Weinstein Model (Physiology and Growth of a Single Tree Through Roots and Foliage); the ROPIS-Chen Model (Physiology and Growth of a Single Tree Through Roots and Foliage); MAESTRO (Tree Crown); AIRPTAEDA (Loblolly Pine Stands); Nutrient Cycling Model; and SILVA (Multiple Species Model). Applications of these models that reference nitrogen oxide contributions to physical effects have not suggested that they play a major role in contributing to forest decline (NAPAP 1990b). Similar conclusions are noted by other researchers who concur that there is no direct evidence connecting acidic deposition with growth declines (US EPA 1994, Lucier 1988, Reams et al. 1990, and Zahner et al. 1989).

Air pollutants are known to alter the diversity and structure of plant communities with the primary effect being on more susceptible members of the plant community in that they can no longer compete effectively for essential nutrients, water, light, and space. Changes in the input of nutrients often result in the decline of sensitive species and enhancement of tolerant species. Many of the relationships central to examining the connections between air deposition and physical effects are uncertain or have only recently been studied in a formal fashion. **Appendix 1 [Inventory of Relevant NO_x Scientific Studies]** summarizes the findings of some studies that have assessed the impact of nitrogen oxide emissions on forests. Within the United States, scientists have suggested that red spruce and fir forests located in high-elevation areas of the Northeast have been adversely affected by acidic clouds (NAPAP 1991b).

Nitrogen enters forest ecosystems in many forms including wet deposition of NH_4, NO_3, and organic N, dry deposition of these forms plus nitric acid vapor, and biological fixation of N_2. Wet and dry deposition inputs enter the forest canopy where they are taken up by trees or by organisms living within the canopy or the phyllosphere. The remaining deposited inputs that are not taken up fall as wet deposition to the forest floor where plants, decomposers, and bacteria compete for it. This competition for nitrogen among heterotrophs, plants, and nitrifying bacteria determines the degree to which vegetation growth is increased or the degree to which incoming nitrogen is retained (US EPA, 1991).

1. Nitrogen Loading Effects

Nitrogen loading effects tend to become significant in forest ecosystems when heterotroph and plant demand for nitrogen are substantially satisfied. This state might be described as nitrogen saturated. Nitrogen deposition is most likely to result in increased vegetation growth given that nitrogen is the most commonly limiting nutrient for growth in forest ecosystems in North America. The extent of the response will depend on the amount of nitrogen deposited, the amount of nitrogen demanded for vegetation, and the competition of heterotrophic soil organisms for this nitrogen. In addition to changes in growth, increased nitrogen deposition can also cause significant changes in tree physiological function, susceptibility to insect and disease attack, and even plant community structure (US EPA, 1991).

Nitrogen deposition to forest ecosystems can affect competitive relationships across plant species and can therefore change species composition and/or diversity (US EPA, 1991). It is important to note that growth responses may not always be deemed desirable especially if they result in changes in species composition. For example, deciduous species have a greater demand for nitrogen per unit biomass produced than do coniferous species. With increased nitrogen deposition, the deciduous species would tend to out compete the coniferous species. There have been some cases where marked changes in species composition have been found as a result of experimental nitrogen additions.

Most forest ecosystems in North America are nitrogen deficient. This fact suggests that increased nitrogen deposition will result in growth increases. Some exceptions might include high elevation areas in the Northeast where it is hypothesized that excess nitrogen deposition has affected the physiological processes of forest ecosystems, disturbing carbohydrate allocation and lowering the tolerance of these systems to drought, frost, or other stress (US EPA, 1991, MacKenzie, 1989). In general, the effects of increased nitrogen inputs on host-pathogens interactions remain speculative, as resistance to insect and disease may be affected either positively (bark beetle) or negatively (pathogenic fungi). When growth is encouraged by excessive levels of nitrogen inputs, forest systems may experience nutrient deficiencies if other nutrients are not in sufficient supply. Physiological and anatomical development can be affected. For example, nitrogen compounds might increase the susceptibility of forest systems to freezing or desiccation in winter. NAPAP conducted research concerning the susceptibility of red spruce to winter injury for high elevation forests in the northeastern portion of the United States. Rain and cloud water in these regions were quite acidic with overall deposition rates being the highest in North America. Scientists tried to link cloud water with freezing of foliage, reduced growth, physical damage to the crown, as well as death of trees. Scientists have established a link between acidic deposition and reduction in cold tolerance but are not certain what components are the principal causal agents or what the damage mechanisms are (NAPAP, 1993).

2. Acidification Effects

Acidification effects on forests are divided into two types, those directly affecting the health of trees and those indirectly affecting tree health through impacts on soil. Direct effects such as significant foliar damage have largely been associated with sulfuric acid rather than nitric acid and only appear pronounced when pH levels are at or below 2.6 (MacKenzie, 1989). The indirect effects via soil acidification within forest systems are likely to be more significant. In certain soils, nitrogen deposition can deplete nutrients by leaching calcium, magnesium, and potassium. These important cations are often replaced by hydrogen ions and together with the mobilization of aluminum can greatly increase soil acidification. Natural weathering in soils can replenish supplies of some nutrients, but if this influence is limited then nutrient imbalances will occur in the forest ecosystems. The mobilization of aluminum is linked with root damage and limited root uptake of calcium and magnesium.

Direct acidification effects include those effects that result from explicit contact between the trees and the atmospheric deposition. Direct effects to the trees might include foliar damage, erosion of epicuticular waxes, foliar leaching, and changes in the physiological functions of tree leaves (Society of American Foresters, 1984). The response of forest ecosystems to atmospheric deposition will vary with the nature and timing of the deposition as well as the type of vegetation exposed. Some species appear to show less tolerance than others and younger trees appear more vulnerable than mature trees.

Experiments with simulated acid rain applications to tree seedlings show foliar damage effects only when the pH drops below 3.5. This pH is far less than that of the average precipitation levels in the United States. Individual trees do come into contact with pH levels much more acidic than the average levels, so the occurrence of such effects cannot be ruled out. Damages to leaf surface waxes and cuticles have also occurred in experimental testing at similarly high levels. Two other types of direct effects are increased permeability of the leaf surface and foliar leaching. Increases in permeability can result in water moving out of the leaf surface and in toxic materials and pathogens moving into the leaf surface. The leaching of substances from foliar tissues can alter the composition of tree tissues and the chemical composition of throughflow and stemflow. Acidification might also be linked with changes in the physiological functioning of tree leaves. Experiments have shown abnormal cell growth under acid rain conditions as well as shifts in photosynthesis or respiration rates. These types of effects are demonstrated at pH levels around 2 to 3 (SAF, 1984).

Although the distinction between direct and indirect effects is somewhat nebulous, indirect acidification effects refer to those that do not involve direct contact between the deposition and the tree organs. Indirect effects might include root effects such as root dieback and toxic effects on tree roots as well as effects on reproductive and regenerative processes.

The sensitivity of forest soils to acid inputs depends on neutralization and buffering capabilities. Salts of weak acids will increase the buffering capacity of soils. The exchangeable acidity of a soil indicates the capacity to buffer acids. The percentage base saturation demonstrates the capacity to neutralize acids. Acidification effects are manifested within soils through nutrient cycling processes. In some cases, nitrogen deposition heightens the leaching of base cations (i.e., calcium, magnesium, and potassium) from soils and increases the concentration of aluminum within soils. When and if base saturation decreases extensively, aluminum soil concentrations can become toxic to plants. Nutrients are supplied to soils by mineral weathering and atmospheric contributions. If this supply is less than that being leached from the soil, nutrient supply will decrease over time and potentially contribute to plant nutrient deficiency and to aluminum toxicity. Fernandez (1989) characterizes soils that might be subject to rapid effects of acidic deposition as those poorly buffered soils with cation exchange capacity < 15 cmol(+)/kg or moderate base saturation soils (i.e., 20 % to 60%) with a pH greater than 4.5. It is likely that thinner soils with low weathering rates will be most vulnerable to soil acidification.

Sensitive forest systems might include systems such as the spruce-fir ecosystems in the northeastern portion of the United States whose soils are naturally acidic, have high levels of organic matter, and very low base saturations in the mineral horizons. The combination of these factors makes these soils susceptible to increases in the amount of soluble aluminum resulting from the deposition of sulfate and nitrate. Soil solution aluminum affects nutrient availability, particularly calcium, which is important to plant growth and development. Increases in the ratio of aluminum to calcium are correlated with decreased growth and reductions in physiological processes as well as the leaching of nutrients from the root zone (NAPAP, 1993).

Concerns have been expressed over the effect of acidic deposition on tree root development. Acidic deposition is often responsible for losses of bases from soils and high levels of mobile aluminum which in turn can decrease fine root biomass and contribute to tree decline. Another effect on roots that can occur in soils with lower pH levels is the increase in levels of trace metals in tree roots and rhizospehere processes. These levels might reach toxic levels as trace metals become mobile. Research on these two root effects has not shown definitive links between deposition quantities and these processes. Changes in reproductive or regenerative processes from acidic deposition are currently being evaluated. Alterations in the timing of development or occurrence of certain processes such as pollen production, fertilization, and seed germination can have broad impacts. Experiments have suggested that seed germination can be inhibited at pH levels below 3. Many seed coats do have buffering capacities however that would limit the likelihood of such inhibitions. Seedling development could potentially be disturbed but experiments have shown mixed results (SAF, 1984).

3. Ozone Effects

The effects of ozone on forest ecosystems have been studied more comprehensively than those of nitrogen oxides. **Appendix 2 [Inventory of Relevant Ozone Scientific Studies]** displays the findings of studies that have attributed reductions in yield as well as changes in species composition to ozone exposure levels. The primary effects of ozone exposure include foliar damage, while secondary effects include reduced root and foliar growth. Visible injury, reduced plant growth, decreased yield, changes in plant quality, and changes in susceptibility to stresses have all been associated with ozone exposure (US EPA 1993).

Major forest species such as ponderosa, loblolly, and slash pine are sensitive to O_3 at concentrations near and above 0.04 to 0.05 ppm. Hogsett et al. (1993) have constructed exposure-response functions for aspen, red alder, black cherry, red maple, and tulip poplar species as well as composite functions for deciduous tree seedlings. Their results suggest that deciduous seedlings and/or fast growing species are more sensitive than evergreen or slower growing seedlings. Other studies on evergreen species have addressed red spruce in the eastern part of the United States, southern pines loblolly and slash) and western conifers (primarily ponderosa pine). Miller et al. (1991) evaluate the contribution of ozone to the compositional changes in the San Bernardino Forest in California. Peterson et al. (1991) discuss similar types of changes in the Sierra Nevada Mountains, as do McLaughlin et al. (1982) in the Appalachian Mountains.

4. Summary of Effects

The results of our literature search on the effects of nitrogen oxide emissionson forest ecosystems illustrate the lack of consensus among scientists concerning the causes of forest decline as well as the uncertainty attributed to several effect pathways. For example, de Steiguer et al. (1990) present the results of a survey of perceptions and estimates by scientists regarding air pollution damages to forests. The results of this survey indicate that sulfuric and nitric acids are of minor concern, except in the case of high-elevation spruce-fir forests, and that ozone is the most destructive pollutant in forest ecosystems.

Potential nitrogen loading effects include species composition changes, reduced foliar nutrition, altered susceptibility of trees to pests and pathogens, altered reproductive or regenerative processes, nutrient stress from leaching of nutrients from foliage and altered photosynthesis, respiration, translocation, and carbon allocation. Possible acidification effects include foliar damage, leaf surface wax and cuticle damage, and changes in physiological functions of tree leaves, root die back, decreased nutrient availability, reductions in pest-resistance and in thermal tolerances, and multiple stress complex sensitivity involving climate, land use changes, and anthropogenic pollutants. Potential effects from ozone exposure increases include visible injury, reduced plant growth, decreased yield, changes in plant quality, and changes in susceptibility to stresses.

C. Linkages to Economic Valuation

Based on this designation of physical effects, impacts on particular service flows provided by forest ecosystems are next explored, with the motivation being the economic valuation of changes in services flows. **Exhibit 2 [Summary of Impacted Service Flows]** displays the likely distribution of impacts from changes in nitrogen oxides emissions. Shaded cells indicate provision of a service by the ecosystem, and shaded cells marked with bolded **Xs** indicate susceptiblity to damage or change.

Changes in the yield or characteristics of forest ecosystems will affect service flow provision[1]. Recreational service flows such as hiking/hunting/camping, birdwatching, and scenic beauty are likely to be affected as tree growth and appearances change. The demand for such recreational activities is linked with forest attributes such as species composition and health. Flood control, nutrient removal, pollutant uptake; and sediment control are likely to be affected by both changes in tree characteristics and soil chemistry. Biodiversity within forest ecosystems is also likely to be affected by changes in tree characteristics and soil chemistry, as well as by shifts in species composition.

To link physical effects with the service flows provided by forest ecosystems, several linkages have to be established. These might include: (1) linkages between deposition of nitrogen oxides and ozone exposure; (2) linkages between nitrogen oxide and ozone exposure and forest yields; (3) linkages between nitrogen oxide and ozone exposure and forest 'quality'; (4) linkages between nitrogen oxide and ozone exposure and chemical/biomass attributes of forests; and (5) linkages between forest yields, forest quality, and other attributes to demands for service flows.

The first linkage will enable potential ozone impacts to be considered. The remaining three linkages represent different pathways to get at changes in services flows. The middle set of linkages would allow for commercial and perhaps changes in recreational flows to be assessed using changes in yield as the relevant metric. The third linkage might allow for changes in recreational flows to, be evaluated using some type of forest quality measure as the metric of changes. The fourth linkage involves the derivation of changes in service flows such as flood control, nutrient removal, pollutant uptake, sediment uptake, and biodiversity preservation. The final linkage addresses the economic valuation of changes in service flows.

Current scientific research does not support the designation of many of these linkages noted above. Some dose-response functions have been developed to describe damages to forest ecosystems from ozone pollution. Hogsett et al. (1993) constructed exposure-response functions for aspen, red alder, black cherry, red maple, and tulip poplar species as well as composite

[1] Commercial service flows from forest ecosystems are directly affected by changes in yield. These flows are not discussed here, as they are not typically considered to be ecosystem effects.

Exhibit 2
Summary of Impacted Service Flows

End Points or Services	Forests	Freshwater Wetlands	Coastal Wetlands	Streams & Lakes	Oceans	Deserts	Grass Lands
Direct Services							
Hiking/ Hunting/ Camping	X	X	X				
Birdwatching/ Wildlife Viewing	X	X	X	X	X		
Scenic Beauty	X	X	X	X	X		
Boating/ Water Activities		X	X	X	X		
Commercial Fishing		X	X	X	X		
Recreational Fishing		X	X	X	X		
Waste Sink							
Indirect Services							
Storm Protection/ Wave Buffering		X	X				
Flood Control		X					
Nutrient Removal Recreation Drinking Water	X	X	X				
Pollutant Uptake Recreation Drinking Water Industrial	X	X					
Sediment Control	X	X	X				
Biodiversity Preservation	X	X	X	X	X		

'X' indicates potential affect from nitrogen oxide controls.

functions for deciduous tree seedlings (see Appendix 2). These functions allow for ozone damages to be expressed in yield terms. This information is relevant to the examination of the second linkage. Physical modeling of the effects germane to the third and fourth linkages have not been well developed.

The final linkage involves placing an economic value on the changes in service flow provision. To comprehensively model the effects of acidification on forest service flows, attempts must be made to capture the behavioral responses of individuals to changes in the components and quality of forest ecosystems. A brief sampling of economic literature suggests that there are two groups of relevant economic studies. These include studies that have specifically addressed the effects of nitrogen oxides or acid deposition and studies that have valued changes in ecosystem components and quality that bear some resemblance to those resulting from nitrogen oxides.

Appendix 3 [Sampling of Economic Studies] provides a description of select economic studies by ecosystem category. Effects on forest service flows have been studied for ozone and acidic precipitation, with many studies focusing primarily on changes in commercial service flows. For example, NAPAP (1991b, 1990b) estimated the changes in consumer and producer surplus from simulated reductions in tree growth within the Southeastern Region of the United States (using the TAMM model). Scenarios were evaluated that assessed a 2% increase in growth and 2% , 5% , and 10% reductions in tree growth. For natural and planted pines, the estimated total change in welfare ranged from $39.3 million for the 2% increase in tree growth to -$108.5 million for the 10% reduction in tree growth. Callaway et al. (1985) evaluated growth reduction scenarios for hardwood and softwood species of 10%, 15%, and 20%. Annual damages from all pollutants ranged from -$270 million for the 10% reduction to -$563 million for the 20% reduction. Crocker (1985b) evaluated the change in welfare associated with a 5% reduction in forest products due to acidic deposition. The annual damage estimate in 1978 dollars was $1,750 million. Haynes and Adams (1992) assessed the economic impact of air pollution to forests using the TAMM model. Losses of 5 to 10% for soft and hard woods were studied, and damages ranged from $1,500 million to $7,200 million. Other studies have assessed changes in the quality characteristics of forests. For instance, Crocker (1985) estimated willingness to pay measures for individuals to prevent ozone damages to the San Bernardino National Forest. Mean additional willingness to pay measures per trip ranged from $2.09 for light damages, $0.66 for moderate damages, and $0.74 for severe damages. Peterson (1987) conducted a similar study to that of Crocker evaluating willingness to pay measures for forest quality effects imposed by ozone. The mean individual willingness to pay to prevent a one step drop on the forest quality ladder was $37.61. Similar studies of changes in forest quality from other types of events (fire and insect damage) are also presented in Appendix 3. Flowers et al. (1985) estimated the willingness to pay for access to different quality forest sites (burned and unburned). Walsh et al. (1989) and Michaelson (1975) looked at changes in consumer welfare associated with insect damage at recreational forest sites.

The difficulty of modeling the entire chain of effects from air pollutants such as nitrogen oxides and ozone is reflected in this sample of economic studies where physical effects have been hypothesized or proxied using photographs. With the exception of the studies that reference changes in yield, there appear to be few studies that lend themselves well to benefits transfer.

3. Freshwater Wetlands

A. Service Flows

Freshwater wetlands provide an array of service flows to individuals. These include direct recreational services such as hiking, hunting, camping, birdwatching, wildlife viewing, boating and other water activities, and commercial and recreational fishing and indirect services such as scenic beauty. Indirect services provided by freshwater wetlands include storm protection, flood control, nutrient removal, pollutant uptake, sediment control, biodiversity preservation, and agricultural pest control. In addition to providing these services to individuals, freshwater wetlands are home to numerous endangered species.

B. Physical Effects

The significance of NO_x emissions in terms of freshwater wetland dynamics will vary greatly across the United States. NO_x emissions may affect service flows from freshwater wetlands through increases in nitrogen nutrient levels, through acidification, and through the effects of ozone on vegetation. NO_x may affect wetlands both through the direct deposition of nitrogen compounds and through the effects of NO_x on the runoff from terrestrial and aquatic systems. Wetland vegetation is commonly nitrogen-limited so changes in nitrogen levels due to NO_x emissions may affect plant populations and indirectly animal populations as a result. Both increases in nitrogen as a nutrient and acidification due to NO_x emissions may have effects on the wetlands nitrogen cycle. These effects may in turn affect wetland plant and animal populations. They may also. affect the emissions and uptake of various gases from wetlands. These gases, specifically CO_2 and N_2O, have been associated with global warming and the destruction of stratospheric ozone.

1. Nitrogen-loading Effects

The effects of increased nitrogen levels in wetlands include an increased threat to endangered plant species, as well as possible large-scale changes in plant populations and community structure. Endangered and threatened plant species are common in wetlands, with wetland species representing 17% of the endangered plant species in the U.S. (US EPA 1991). These plants are often specifically adapted to low nitrogen levels, for example, isoetids (Boston 1986 as cited by US EPA 1991) and insectivorous plants (Moore et al. 1989). In eastern Canadian wetlands, nationally rare species are most common in infertile sites (Moore et al. 1989, Wisheu and Keddy 1989 as cited by U.S. EPA 1991). When nitrogen levels increase, other species adapted to higher levels of nitrogen may competitively displace the endangered and threatened species. Thus, NO_x emissions which increase nitrogen levels in nitrogen-poor wetlands may increase the danger to threatened and endangered species. More common species which thrive in nitrogen-poor wetlands may also become less common.

By changing competitive relations between plant species, increased nitrogen inputs may broadly affect plant populations and therefore community structure in certain wetlands. Many northern nitrogen-poor bogs are dominated by *Sphagnum* species. These species capture low levels of nitrogen from precipitation. Increased nitrogen levels may directly harm *Sphagnum* as well as causing increased nitrogen to be available to vascular plants which may outcompete the *Sphagnum* (Lee & Woodin 1988, Aerts et al. 1992). Studies in Great Britain have documented large declines in *Sphagnum* moss populations as a result of atmospheric pollution (Ferguson et al. 1984, Lee et al. 1986, both as cited by US EPA 1991). However, in these cases, it is difficult to distinguish the community effects of NO_x emissions from the effects due to other pollutants (Lee et al. 1990). Increased nitrogen delivery levels may also affect the carbon balance of carbon-accumulating peatlands. Increased nitrogen in plant matter appears to cause increased decomposition of dead plant matter, which could affect the carbon accumulation of peatlands and therefore the global carbon balance (Clymo & Hayward 1982 as cited by Lee & Woodin 1988; Aerts et al. 1992).

Increases in nitrogen levels due to NO_x emissions will be most significant in wetlands which are extremely nitrogen-limited and which receive small amounts of nitrogen naturally relative to increased nitrogen receipt due to NO_x emissions. Ombrotrophic bogs are bogs which receive little surface water runoff and receive most of their nutrient and water inputs through precipitation. These ombrotrophic bogs may receive one to two orders of magnitude less nitrogen than salt marshes and other freshwater wetlands (10 kg N/ha/yr vs. 53, 76, and 740 kg N/ha/yr) which receive nitrogen through ground, surface, and tidal waters (US EPA 1991). As total bulk deposition of nitrogen has been estimated to be at least 5.5-11.7 kg N/ha/yr (US EPA 1991), changes in NO_x emissions would most likely impact wetlands such as raised peat bogs. Logofet and Alexandrov (1984) suggest that a treeless oligotrophic (nutrient-poor) bog may undergo succession to a forested bog as a result of the input of as little as 7 kg N/ha/yr. In contrast to these ombrotrophic bogs, the effects of NO_x levels on wetlands which receive significant surface water runoff will be greatly affected by the nitrogen retention of the surrounding watershed.

2. Acidification Effects

Acidification may affect the nitrogen cycle in wetlands in numerous ways. The acidic and anaerobic nature of wetland soils results in losses of nitrous oxide (N_2O) and nitrogen (N_2) to the atmosphere, through bacterial processes. Decreasing pH appears to increase losses of N_2O relative to N_2, which may affect global warming and stratospheric ozone (Focht 1974 as cited by US EPA 1991). Acidification also decreases nitrification, the conversion of NH_4^+ to NO_3^-, as well as affecting other processes within the nitrogen cycles of wetlands (US EPA 1991). It is unclear how these results of acidification on the nitrogen cycle would affect service flows or how NO_x reductions would limit possible damages.

3. Summary of Effects

Potential nitrogen-loading effects include changes in species composition within wetlands and in plant decomposition. Potential acidification effects include changes in nitrogen cycling and possible resulting changes in the contribution of wetlands to gases affecting global warming and stratospheric ozone. It appears that nitrogen loading effects may be more directly relevant in terms of ecosystem effects.

C. Linkages to Economic Valuation

The potential effects on particular service flows provided by freshwater wetland ecosystems are next explored, with an emphasis placed on the economic valuation of changes in services flows. **Exhibit 2 [Summary of Impacted Service Flows]** displays the likely distribution of impacts from changes in nitrogen oxides emissions. Shaded cells indicate provision of a service by the ecosystem, and shaded cells marked with bolded **Xs** indicate susceptiblity to damage or change.

Changes in the species composition of flora in freshwater wetlands will potentially affect service flow provision. If changes in flora involve endangered species, there are likely to be significant losses associated with the loss of such plants and in the biodiversity of wetland ecosystems. Recreational service flows such as hiking/hunting/camping, birdwatching/wildlife viewing, scenic beauty, boating or water activities, and fishing might also depend on the species composition of the wetland ecosystem. Flood control, storm protection, nutrient removal, pollutant uptake, and sediment control may vary with the nutrient dynamics of wetlands as well as the species comprising the ecosystem.

To link these effects with service flow provision, the following connections must be established: (1) linkage between nitrogen oxide deposition and the nutrient dynamics of wetlands; (2) linkage between wetland nutrient dynamics and species composition; (3) linkage between species composition and quality of service flow provision; and (4) linkage between demand for service flows and quality of provision. The first and second linkages emphasize the designation of physical effects, while the third and fourth establish mechanisms to link these effects with economic valuation.

Current science does not well support the identification of the first two linkages. There is considerable variation in the species composition of freshwater wetlands across the United States, and it is difficult to predict the succession of species under different circumstances. In addition, if benefits from NO_x reductions are to be received, the effects of NO_x emissions on wetlands must be reversible or NO_x reductions must interrupt continuing effects. Little evidence is available concerning the reversibility of the effects described above.

The third and fourth linkages also appear difficult to establish. It is troubling to consider the impact of different species on service flows, for individual species are rarely singled out in analyses of service flow provision. To assess the effects of nitrogen oxides emissions on freshwater wetlands requires that the interdependencies between wetland composition and service flow provision be delineated and that variations in service flow provision be valued. A sampling of economic studies presented in Appendix 3 suggests that studies have attempted to value the direct and indirect services provided by freshwater wetlands. This suggests that the final linkage has been addressed previously. For example, Whitehead and Blomquist (1991) used the contingent valuation method to derive household values for natural wetlands in Kentucky. Individuals were shown pictures of wetlands in natural and commercially developed states and were educated about the various services provided by wetlands. The annual per household value (preservation value) of the Clear Creek wetland in western Kentucky estimated ranged from $5 to $17. Likewise, Raphael and Jaworski (1981) estimated per acre values for waterfowl hunting ($49.59), nonconsumptive recreation ($244.63), water quality and nutrient uptake ($930-$1,124), and water supply services ($10.71) provided by Michigan wetlands. These authors used recreational expenditures as a proxy. Thibodeau and Ostro (1981) examined the value of recreational fishing and hunting service flows provided by wetlands. Using the contingent valuation method, these service flows were estimated to be worth $281.30 per acre. Hamm (1991) explored the value of nonconsumptive recreation, education, water quality and nutrient uptake services provided by South Carolina wetlands. Hamm (1991) employed the contingent' valuation method and estimated these services to be worth approximately $0.5 to $1.2 million, $0.08 million, and $4.9-$5.7 million respectively. In addition, Bowker and Stoll (1988) used the contingent valuation method to derive existence value estimates for wetlands located in an Arkansas Wildlife Refuge. Per person values ranged from $28.17 to $73.65. Tilton et al. (1977) used avoided cost methods to value the water quality and nutrient uptake services provided by freshwater marshes. Their estimates ranged from $47,500 to $81,000. Similarly, Wharton (1970) used the avoided cost/damage method to value the water supply services provided by wetlands. An estimate of approximately $800,000 was derived. Amacher et al. (1989) used an hedonic price regression analysis to derive the preservation value of Michigan wetlands. The analysis revealed a value of approximately $38.99 per acre.

In general, the economic studies have not concentrated on changes in wetland quality from physical effects modeling. Rather, the studies have measured total values of existing services or imposed scenarios in contingent valuation surveys. Direct services have been valued using the contingent valuation method, travel cost method or recreational expenditures as a proxy, while indirect services have been examined using avoided cost techniques. In terms of benefits transfer, many of the studies lend themselves more closely to circumstances where catastrophic changes in the wetland have occurred unlike the species composition changes of interest here.

4. Coastal Wetlands

A. Service Flows

Coastal wetlands provide an array of service flows to individuals. These include direct recreational services such as hiking, hunting, camping, birdwatching, wildlife viewing, boating and other water activities, and commercial and recreational fishing and indirect services such as scenic beauty values. Indirect services provided by coastal wetlands include storm protection, nutrient removal, pollutant uptake, sediment control, biodiversity preservation, and agricultural pest control. In addition to providing these services to individuals, coastal wetlands are home to numerous endangered species.

B. Physical Effects

The effects of nitrogen oxide emissions on coastal wetlands rests on the extent to which the nitrogen cycle dynamics are affected. Acidification effects are not likely to occur in such systems because of several factors including the frequency of in and out fluxes of water and the extent of vegetative buffers in some cases.

Nitrogen Loading Effects

Coastal wetlands tend to ‘receive large nitrogen inputs naturally. In terms of nitrogen sources, the most important contributor to coastal wetlands are tidal waters and in some cases ground water. Deposition tends to play somewhat of a lesser role in coastal wetlands systems. The nitrogen cycle in coastal salt marsh and fens wetland ecosystems is open, with significant exchanges of nitrogen with other ecosystems. Similar to freshwater wetlands, increased nitrogen inputs will increase productivity and alter the competitive relationships between species in coastal wetlands (US EPA 1991). Although increased nitrogen does increase coastal wetland productivity, studies showing this have used large quantities of nitrogen (100-3000 kg N/ha/yr) (US EPA 1991). Therefore, limited changes in NO_x emissions may have little effect on coastal wetland productivity, especially if these wetlands are buffered from effects by the retention of nitrogen by terrestrial ecosystems.

C. Linkages to Economic Valuation

Attention is next devoted to understanding the potential effects on particular service flows provided by coastal wetland ecosystems, with an emphasis placed on the economic valuation of changes in services flows. **Exhibit 2 [Summary of Impacted Service Flows]** displays the likely distribution of impacts from changes in nitrogen oxides emissions. Shaded cells indicate provision of a service by the ecosystem, and shaded cells marked with bolded **Xs** indicate susceptiblity to damage or change.

Changes in the species composition of flora in coastal wetlands will potentially affect service flow. provision. If changes in flora involve endangered species, there are likely to be significant losses associated with the loss of such plants and in the biodiversity of wetland ecosystems. Recreational service flows such as hiking/hunting/camping, birdwatching/wildlife viewing, scenic beauty, boating or water activities, and fishing might also depend on the species composition of the wetland ecosystem. Flood control, storm protection, nutrient removal, pollutant uptake, and sediment control may vary with the nutrient dynamics of wetlands as well as the species comprising the ecosystem.

To link these effects with service flow provision, the following connections must be established: (1) linkage between nitrogen oxide deposition and coastal wetland nutrient dynamics; (2) linkage between coastal wetland nutrient dynamics and species composition; (3) linkage between species composition and quality of service flow provision; and (4) linkage between demand for service flows and the quality of the service flows provided. The first and second linkages focus on describing the physical effects, whereas the latter linkages address the economic valuation of human responses to these physical effects.

Current science does not well support the identification of the first two linkages. The sources that contribute to the nutrient budgets of coastal wetlands are varied and so then will the impact of changes in pollutant emissions. Even if changes in species composition could be predicted, the resulting effects on service flow provision are quite difficult to perceive. Similar to the case of freshwater wetlands, little research has focused on the reversibility of such changes in species composition.. The final linkages in examining the impacts of changes in nitrogen oxide emissions involve attaching monetary values to variation in service flows. As noted in the discussion of freshwater wetlands, several studies have attempted to place values on the service flows provided by wetlands. A sampling of economic studies are presented in Appendix 3. The majority of studies appear to focus on freshwater wetlands rather than coastal wetlands. Some exceptions to this trend found in the sampling include Bergstrom et al. (1990) and Costanza et al. (1989). Bergstrom et al. (1990) study that evaluates the on-site recreational services provided by coastal wetlands along the Gulf of Mexico coast of Louisiana. This study employed the contingent valuation method. The estimated annual average consumer surplus per acre was approximately $8.42 (1987 $). Aggregate consumer surplus on an annual basis neared $118 million (1987 $). Costanza et al. (1989) used an energy analysis approach that looks at the amount of energy captured by natural ecosystems as an estimate of their potential to do useful work for the economy.

Once again, the economic studies do not necessarily correspond to the physical effects of interest. The studies have not concentrated on particular compositional changes in wetland quality but have measured total values of existing services or imposed scenarios in contingent valuation surveys. In terms of benefits transfer, many of the studies lend themselves more closely to circumstances where catastrophic changes in the wetland have occurred unlike the species composition changes of interest here.

5. Streams and Lakes

A. Service Flows

As displayed in **Appendix 1 [Inventory of Ecosystem Services],** stream and lake ecosystems provide a tremendous variety of service flows to individuals. These include direct recreational services such as birdwatching, wildlife viewing, boating and water activities, and commercial and recreational fishing and direct nonuse services such as scenic beauty and waste sink services. Indirect services provided by lake and stream ecosystems include flood control, sediment control, and biodiversity preservation.

B. Physical Effects

Sources of nitrogen within aquatic systems of streams and lakes include: point source and non-point source pollution, biological fixation of nitrogen, and deposition of nitrogen oxides. The estimation of the relative contributions of nitrogen is made difficult by the large variety of compounds found in air, deposition, watersheds, and surface waters, and by the myriad pathways through which nitrogen can be cycled in terrestrial and aquatic systems. There are three categories of effects of nitrogen deposition on aquatic ecosystems. These include acidification, eutrophication, and direct toxicity effects. The current high capacity of forests to retain nitrogen has resulted in limited study of nitrogen deposition. Nitrogen saturation of forest watersheds will result in higher supplies of nitrogen to surface waters. The level of nitrogen released to surface waters exhibits considerable seasonal variation, as demands for nitrogen within forest systems vary (NAPAP, 1993). In aquatic systems, nitrogen additions do not usually stimulate growth as systems are rarely nitrogen limited.

1. Nitrogen Loading Effects

The primary effect of increased nitrogen loadings in aquatic systems is eutrophication. Eutrophication is defined as the state of nutrient enrichment or the process by which marine systems progress to a state of eutrophy. The link between eutrophication and nitrogen deposition depends on the extent to which the productivity of system is limited by nitrogen availability and the relative importance of nitrogen deposition compared to other sources of nitrogen within the ecosystem. In freshwater systems, phosphorous, rather than nitrogen, is often the limiting nutrient. When nitrogen limitation does occur, it is often short-lived. Nitrogen fixation by species can often remedy the situation naturally (NAPAP, 1990b).

Nitrogen limitation often occurs in lakes with low concentrations of both phosphorous and nitrogen. This includes lakes within the western and northeast portions of the United States. In general, however, the potential for nitrogen deposition to contribute to eutrophication in freshwater lakes and streams is quite limited. If freshwaters are nitrogen limited, excess nitrogen deposition can result in tremendous damage to fish stocks. This damage occurs through the effect of nutrient levels on algae growth. The growth of algae plants is encouraged by higher nitrogen levels. These plants block sunlight from reaching vegetation in the waters that act as food and habitat for numerous aquatic species. When the plants die and fall to the bottom, they contribute to reduced oxygen levels in waters as decomposing bacteria increase their demand for oxygen leaving less for other species.

2. Acidification Effects

Two types of acidification effects are designated within freshwater aquatic ecosystems, Chronic effects involve long-term or permanent reactions, while, episodic effects involve event-based responses that vary with seasons, temperature, or some other factor. For example, during the Summer and Fall, demand for nitrogen by watersheds is high, while supply rates are low and hence there is a low probability of nitrogen leaching unless nitrogen saturation occurs. Current science shows no evidence of chronic acidification problems in the United States. Episodic responses have been noted in the following areas: Northeast, Mid-Atlantic, Mid-Atlantic Coastal Plain, Southeast, Upper Midwest, and Western regions. Lakes appear to be more sensitive than streams to nitrogen deposition because of their greater water retention periods. Episodic events are often linked with seasonal phenomena such that the highest impact of nitrogen in, surface water acidification occurs during high-flow seasons and periods of snow-melt (NAPAP, 1991b, 1990a). EPA conducted an episodic response project in 1993 that examined the effects of episodic acidification in 13 streams in the Adirondack and Catskill Mountains in New York and the Northern Appalachian Plateau in Pennsylvania. The results of this project suggest that nitrate deposition can contribute to the occurrence of episodes and their adverse effects on fish populations. (Canada-United States Air Quality Committee, 1994).

Direct biological effects through physiological stress and toxicity processes occur as well as indirect effects that manifest themselves through food availability and predation. Although biological effects of acidification impact algae, zooplankton, benthic invertebrates, fish, amphibians, and water fowl, most research has concentrated on fish populations (NAPAP, 1990a). **Exhibit 3 [Summary of Biological Changes with Surface Water Acidification]** presents descriptions of the biological effects of acidification at different pH increments.

The National Surface Water Survey estimated that 4 percent of the lakes and 8 percent of the streams in acid-sensitive regions of the United States were chronically acidic (acid neutralizing capacity (ANC) of 0 or less. ANC is a measure constructed to reflect a water body's sensitivity or response to acidification.[2] Acid-sensitive waters fall in the following regions: Mid-Appalachians, Adirondacks, New England, the Mid-Atlantic Coastal Plain, Florida, and the eastern portion of the Upper Midwest. In addition to these chronic areas, episodic events are likely to occur in waters with even higher ANC levels. Watersheds that show signs of nitrogen saturation tend to have long histories of elevated rates of nitrogen deposition, large pools of soil nitrogen, and maturing forests (NAPAP 1993). **Exhibit 4 [National Surface Water Survey - Lake Acidification]** and **Exhibit 5 [National Surface Water Survey - Stream**

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[2] ANC is expressed in units of micro equivalents per liter (μeq/l), where an equivalent is the capacity to neutralize one mole of H^+ ions.

Exhibit 3
Summary of Biological Changes with Surface Water Acidification[1]

ph decrease	general biological effects
6.5 to 6.0	Small decrease in species richness of phytoplankton, zooplankton, and benthic invertebrate communities resulting from the loss of a few highly acid-sensitive species, but no measurable change 'in total community abundance or production. Some adverse effects (decreased reproductive success) may occur for highly acid-sensitive fish species (i.e., fathead minnow, striped bass).
6.0 to 5.5	Loss of sensitive species of minnows and dace, such as blacknose dace and fathead minnow;. in some waters decreased reproductive success of lake trout and walleye. Visual accumulations of filamentous green algae in the littoral zone of many lakes and in some streams. Distinct decrease in the species richness and change in species composition of the phytoplankton, zooplankton, and benthic invertebrate communities. Loss of a number of common invertebrate species from the zooplankton and benthic invertebrate communities, including zooplankton species such as *Diaptomus silicis, Mysis relicta, Epischura lacustris;* many species of snails, clams, mayflies, and amphipods, and some crayfish

Exhibit 3
Summary of Biological Changes with Surface Water Acidification[1]

5.5 to 5.0	Loss of several important sport fish species, including lake trout, wallyeye, rainbow trout, and smallmouth bass; as well as additional non-game species such as creek chub Further increase in the extent and abundance of filamentous green algae in lake littoral areas and streams Continued shift in the species composition and decline in species richness of the phytoplankton, periphyton, zooplankton, and benthic invertebrate communities; decreases in the total abundance and biomass of benthic invertebrates and zooplankton may occur in some waters Loss of several additional invertebrate species common in olitrophic waters, including Daphnia *galeata mendotae, Diaphanosoma leuchtenbergianum, Asplancha priodonta;* all snails, most species of clams, and many species of mayflies, stoneflies, and other benthic invertebrates Inhibition of nitrification

Exhibit 3
Summary of Biological Changes with Surface Water Acidification[1]

5.0 to 4.5	Loss of most fish species, including most important sport fish species such as brook trout and Atlantic salmon; few fish species able to survive and reproduce below pH 4.5 (i.e., central mudminnow, yellow perch, and largemouth bass) Measurable decline in the whole-system rates of decomposition of some forms of organic matter, potentially resulting in decreased rates of nutrient cycling Substantial decrease in the number of species of zooplankton and benthic invertebrates and further decline in the species richness of the phytoplankton an diperiphyton communities; measurable decrease in the total community biomass of zooplankton and benthic invertebrates in most waters Loss of zooplankton species such as *Tropocyclops prasinus mexicanus, Leptodora kinditi,* and *Conochilis* unicornis; and benthic invertebrate species, including all clams and many insects and crustaceans Reproductive failure of some acid-sensitive species of amphibians such as spotted salamanders, Jefferson salamanders, and the leopard frog

Notes:
[1] Reproduced from NAPAP (1990a) SOS-T 14, page 13-173.

Acidification] display descriptive statistics for several regions within the United States. The ranges of pH, ANC, and dissolved organic carbon (DOC) in lakes are presented by region. Florida has the highest percentage (35%) of acidified lakes (ANC < 0), followed most closely thereafter by the Adirondack Mountain (14%) and Upper Midwest (3%) Regions. The percentages of streams by ANC category are presented for different regions. Florida and Mid-Atlantic sub-regions appear to harbor more sensitive stream waters. In many regions, there was not sufficient data to provide for such calculations.

Land use characteristics such as shallow, coarse textured soils, land in rock outcrop, soils with low base saturation, soils containing rock that weathers slowly, and rapid delivery of water during summer snow melt (NAPAP, 1993) will increase the sensitivity of certain aquatic systems to acidification effects. Acidification basically has two effects: modifying the acid/base chemistry of the system and inducing biological effects. As was shown in **Exhibit 3 [Summary of Biological Changes with Surface Water Acidification],** species respond differently to lower pH levels. For fish, occurrence and successful reproduction vary considerably with pH level. Decreases in pH (below 6.0-6.5) and increases in the concentration of inorganic monomeric aluminum (above 30-35 micrograms per liter for most sensitive organisms) can cause quite adverse effects (NAPAP, 1993). Biological effects on fish from toxic situations such as high NH, conditions include lesions in gill tissue, reduced rates of trout fry, reduced fecundity, increased egg mortality, and increased susceptibility of fish to other diseases as well as a variety of pathological effects in invertebrates and aquatic plants (US EPA, 1991). **Exhibits 4 [Estimated Critical pH Values]** and **5 Fish and pH Sensitivity]** indicate the range of pH values that result in certain adverse effects. Ecosystem-level processes such as decomposition, nutrient cycling, and productivity are affected at chronic pH levels below 5.0-5.5. Net changes in communities often occur with less tolerant species being replaced by more tolerant species (Baker 1985).

C. Summary of Physical Effects

Potential effects from increased nitrogen loadings include eutrophication impacts where excess nitrogen deposition can result in tremendous damage to fish stocks, as marked increases in algae growth disturb the balance of oxygen demands within the system, cloud water, and block sunlight. The combination of these influences imposes stresses on bottom-dwelling fish and other plant species. These effects are not very common in freshwater lake and stream ecosystems. Possible acidification effects include effects such as recruitment failure, population loss, fish kill, adult mortality, embryo mortality, fry mortality, reduced production of viable eggs, and reduced growth amongst fish populations. In addition, long term changes in pH levels can result in species composition shifts with the more acid tolerant species outcompeting those that are less acid tolerant.

3. Linkages to Economic Valuation

Based on this designation of physical effects, impacts on particular service flows provided by forest ecosystems are next explored, with the motivation being the economic valuation of changes in services flows. **Exhibit 2 [Summary of Impacted Service Flows]** displays the likely distribution of impacts from changes in nitrogen oxides emissions. Shaded cells indicate provision of a service by the ecosystem, and shaded cells marked with bolded **Xs** indicate susceptiblity to damage or change. Changes in the water chemistry of streams and lakes are likely to impact the provision of service flows by these ecosystems. These changes will occur from direct deposition of nitrogen oxides to these waters as well as runoff from terrestrial ecosystems (which varies with soil chemistry). As the species living in these waters and the quality of the waters change, the provision of service flows are likely to be altered. Recreational service flows such as birdwatching/wildlife viewing, scenic beauty, boating and other water activities, and fishing and biodiversity services might be affected by such changes, in species and water quality.

To assess the potential effects of nitrogen oxide reductions on lake and stream ecosystem. service flows; several linkages must be established. These linkages include: (1) linkage between nitrogen oxide emissions and water chemistry (i.e., pH or ANC) of lakes and streams; (2) linkage between nitrogen oxide emissions and eutrophication processes; (3) linkage between water chemistry and fish stocks and/or water quality; (4) linkage between eutrophication and fish stocks and/or water quality; and (5) linkages between the demand for service flows and fish stocks and water quality.

As part of the NAPAP research effort, several models were developed to simulate changes in water chemistry, but many of these models (i.e., MAGIC) are quite data intensive (i.e., water body specific data) and do not lend themselves well to a national analysis. In addition, very few of these models specifically recognize the contributions of nitrogen oxides to changes in water chemistry. Many of these models were designed to assess the contributions of sulfates rather than nitrates. Both steady-state and dynamic models (ETD, ILWAS, and MAGIC) are discussed in NAPAP (1990a). The EPA Episodic Response Project might provide more valuable information about the role of nitrates. Overall, the research tendency to favor sulfates complicates the modeling of the first linkage. Similarly, the second linkage might be proxied in well-studied areas such as the Chesapeake Bay, but it is unlikely that the contribution of NO_x emissions to eutrophication might be easily specified.

The third and fourth linkages involve predicting the response of species and vegetation to changes in water chemistry. As part of NAPAP, toxicity models were also developed to allow for the linkage between pH and species existence to be established. An acidic stress index (ASI) is developed that indicates the probability that a species will be absent from a particular water body. **Exhibit 2 [Inventory of Relevant NO_x Scientific Studies]** includes several regional case studies that assessed acidification and its impacts on aquatic life. These case studies essentially were designed to focus on the third and fourth linkages listed above. It appears that some type of predictions of species existence might be made if water chemistry and other data are collected for regional waters.

The final step in assessing the effects of nitrogen oxide emissions on lake and stream ecosystems consists of contemplating the impact of physical effects on human behavior. To place monetary values on the changes in service flows requires an understanding of how the demand for certain services will change or how the provision of certain services will be altered. Appendix 3 presents a sampling of economic studies on fishery ecosystems and acidification. It is important to note that these studies tend to be site specific. For example, NAPAP(1991b) estimated welfare changes of the cold-water recreational anglers in New York, New Hampshire, Vermont, and Maine. Random utility and travel cost models were applied to evaluate the impacts of a 50% reduction in acidic deposition, no change in deposition, and a 30% increase in deposition. Welfare changes were calculated as differences between total welfare in 1990 and 2040. Results for the three scenarios from the random utility and hedonic travel cost model were $14.7 million and $4.2 million for the 50 % reduction, -$5.3 million and -$27.5 million for the no change in deposition, and -$10.3 million and -$97.7 million for the 30% increase in deposition. Similarly, Mullen and Menz (1985) assessed the welfare implications of acidification on angling sites in the Adirondacks Mountain Region. Angling sites were turned off when pH levels dropped below 5. The travel cost modeling estimate of the loss in net economic value was approximately $1.1 million. Likewise, Callaway et al. (1990) also examined the impact of acidification on recreational fishing service flows. Scenarios that resulted in 3.2% and 10% increases in fishable acreage as well as combined acreage and catch rate increases of the same size were evaluated. The values for the acreage scenarios ranged from $700,000 to $4.6 million, while the acreage and catch rate scenarios produced estimates that ranged from $4.8 million to $12.0 million. IN addition, Morey and Shaw (1990) completed similar research that linked changes in fish stocks and catch rates with changes in acidic deposition. Reductions in catch rates of 5% , 25 % , and 50% generated mean annual expected CCVs of $1.91, $8.68, and $15.51. In terms of benefit transfer, several of these economic studies seem relevant as they focus on changes in fish stock. Although, it might be problematic to use these results on a large scale, for the studies address impacts in the Northeastern region of the country only.

Exhibit 4
National Surface Water Survey - Lake Acidifictaion[1]

Study Area	Sample Sue	pH	ANC (μ eq/ liter)	DOC (μ M)	ANC-% (< 0) (< =200)	pH - % (< 5) (< 6)
Adirondack Mountains	104	5.4-6.9	5-136	244-445	18.4 85.9	14.4 35.8
Maine	185	6.6-7.2	69-252	272-653	1.5 69.4	0.5 5.1
Upper Midwest	426	6.3-7.2	55-404	400-908	5.4 57.8	2.9 14.5
Southern Blue Ridge	54	6.6-6.9	89-220	83-117	0 70.4	0 1.2
Florida	31	4.8-6.4	-10-49	328-1130	46.0 87.1	34.8 61.7
West	393	6.8-7.3	48-160	51-159	0 83.0	0 1.2

[1] Source: Baker et al. (1991). Tables 17.3 and 17.4. p. 579-580. Data are from the National Surface Water Survey data base.

Exhibit 5
National Surface Water Survey - Stream Acidification[1]
Percentage of Streams by ANC category

Study Area	Sample Size	ANC (< 0)	ANC (0-50)	ANC (50-100)	ANC (100-200)	ANC (>200)
Mid-Atlantic						
Coastal Plain	43	13.5	25.5	6.2	23.5	31.4
Piedmont	22	4.2			8.4	87.4
Valley and Ridge	48	7.2	7.8	4.5	32.6	47.9
Appalachian Plateau						
	90	6.6	24.8	12.9	19.0	36.8
Interior Southeast						
Piedmont						
Blue Ridge	38			9.7	22.8	67.6
Valley and Ridge	60		8.3	36.2	26.2	29.4
Appalachian Plateau	15		6.7		8.7	84.6
Ozark/Ouachita	17	5.6	5.6	18.6	11.3	58.9
Gulf Coastal Plain						
	40		2.1	24.1	46.2	27.5
	18		32.4	43.1	10.7	13.8
Florida						
Panhandle	18	27.7	48.4	14.4		9.6

[1] Source: NAPAP (1990a). Volume II. Report 9.

Exhibit 6
Estimated "Critical" Ph Values[1]

Species	pH Value	Number of Studies Used
Mudminnow	4.6	5
Yellow Perch	4.8	16
Brown Bullhead	4.9	15
Pumpkinseed Sunfish	4.9 ,	10
Largemouth Bass	5.0	12
Northern Pike	5.1	12
Brook Trout	5.2	53
White Sucker	5.2	26
Rock Bass	5.2	9
Golden Shiner	5.2	15
Arctic Char	5.2	3
Atlantic Salmon	5.3	14
Brown Trout	54	17
Creek Chub	5.4	14
Rainbow Trout	5.5	15
Smallmouth Bass	5.5	18
Lake Trout	5.6	28
Walleye	5.9	13
Redbelly Dace	6.0	8
Slimy Sculpin	6.0	3
Common Shiner	6.1	11
Fathead Minnow	6.1	7
Blacknose Dace	6.3	7
Bluntnose Minnow	6.3	7
Blacknose Shier	6.3	5

Notes:
[1] This table was adapted from Table 4.2 presented on page 99 of Baker and Christensen (1991).

Exhibit 7
Fish and pH Sensitivity

Fish	Recruitment Failure	Population Loss	Population Absence
Rainbow Trout			4.9-5.0 4.7-5.7 5.4
Lake Trout	6.9 5.5 5.5-5.6 5.2-5.5	5.4 5.2 5.3-55 5.2-5.3 5.0-5.2 5.2-5.8	4.9 4.4 5.1
Brook Trout	4.9-5.4	5.0 4.3-5.0 4.7-5.0 5.1-5.6 4.6-5.6 4.6-5.1	5.1 4.9 5.8 4.6-4.7 5.1-5.3 5.6 5.0 4.6
Brown Trout	4.9-5.3 5.0	5.4-5.6 4.8-6.0 4.6-5.3 5.0-5.1	4.4-4.8 4.6
Atlantic Salmon		5.0-5.1 4.7-5.0	5.3
Creek Chub	4.8-4.9 4.9-5.0 4.9-5.5 4.7-5.2	5.4 4.7-5.0	5.2 5.1 6.0 5.9 5.6 5.0 4.6 5.7
Blacknose Dace		6.2 5.7-6.1	6.5 5.6
Arctic Char	5.2		
Northern Pike	4.9-5.0 4.4-4.9	4.7-5.2 4.6	5.1 4.2-4.3 4.0 5.2 5.6 5.9
Yellow Perch	4.4 4.5-4.7 4.4-5.0	< 5.1 < 4.7	4.8-4.9 4.6 4.2-4.3 4.4 5.5-5.7 4.5

Exhibit 7
Fish and pH Sensitivity

Fish	Recruitment Failure	Population Loss	Population Absence
Walleye	5.9 5.5-6.0	5.5 5.2-5.9 5.2-5.8	5.2 5.5
Fathead Minnow			5.1 5.5 6.3
Mud Minnow			4.8-4.9 4.0 4.2 4.5
Rock Bass	4.9-5.0 4.7-5.2 4.8-5.0	4.7-5.2	5.2 4.2-4.3 5.2 4.6
Smallmouth Bass	5.5-6.0 5.0	4.4 6.0	5.6 4.9 4.4 5.2 5.5-5.7 7.0
Golden Shiner	5.2	< 5.1 4.6-4.8 4.7-5.0	4.8-4.9 5.1 5.5 5.2 5.0-5.1 4.7 5.9 4.5 5.1 4.6-4.8
Largemouth Bass	5.2 4.1-4.5	4.9-5.4 < 3.7	4.9 4.4 4.6 4.1-4.5 4.7 5.0
White Sucker	4.8-4.9 4.9-5.0 4.9-5.5 4.7-5.2	4.5-4.8 4.9-5.1 5.4-5.7 4.7-5.2	4.8-4.9 4.2-4.3 4.6-4.7 5.4 4.9 5.4-5.5 5.2 4.6 5.5

Exhibit 7
Fish and pH Sensitivity

Fish	Recruitment Failure	Population Loss	Population Absence
Brown Bullhead	4.9 4.7-5.2	< 5.1 4.2 - 4.5 4.7 - 5.2	4.8-4.9 4.6-4.7 4.5 5.4 4.5
Pumpkinseed Catfish		4.8-4.9 4.7-5.2	4.8-4.9 4.6 4.2-4.3 5.4 4.9 5.5-5.7 4.6 4.9
Common Shiner			5.6 5.1 5.7 5.3 6.0 6.2 5.4 4.9 6.1

Source: Baker et al (1985) and Baker and Christensen (1991).

6. Oceans and Estuarine Ecosystems

A. Service Flows

Ocean waters provide numerous services to individuals. Direct recreational services include birdwatching and wildlife viewing, boating and water activities, commercial fishing, and recreational fishing. Direct nonuse services include scenic beauty, whereas indirect services offered by ocean waters include biodiversity preservation.

B. Physical Effects

1. Eutrophication Effects

In contrast to the situation with lakes, there does appear to be some evidence of nitrogen limitation in many estuarine and coastal marine ecosystems. When nitrogen limited ecosystems receive nitrogen depositions, the potential for eutrophication dynamics to be initiated is high. While eutrophication affects a variety of species, it can particularly prove damaging to fish stocks. This damage occurs through the effect of nutrient levels on algae growth. Higher nitrogen levels incite increased algae growth. As these tiny plants accumulate and fill waters, they block sunlight which grasses and other forms of vegetation depend on. This effect has repercussions throughout the ecosystem, as vegetation serves as an important source of food and habitat for numerous aquatic species. When these algae die and fall to the lower sediments, the decomposing bacteria increase their demand for oxygen leaving less for other species including bottom-dwelling fish, shellfish, and other plant species.

Eutrophication is defined as the state of nutrient enrichment or the process by which marine systems progress to a state, of eutrophy. The link between eutrophication and nitrogen deposition depends on the extent to which the productivity of system is limited by nitrogen availability and the relative importance of nitrogen deposition compared to other sources of nitrogen within the ecosystem. Nitrogen deposition is typically not thought of as a primary source of nitrogen for a coastal ecosystem. However in some cases such as the Chesapeake Bay, nitrogen deposition appears to play a significant role in terms of nitrogen entering the ecosystem. The Environmental Defense Fund (Fisher et al. 1988) suggests that nitrogen deposition from the burning of fossil fuels accounts for one quarter of the nitrogen entering the Chespeake Bay. Other models suggest that nitrogen deposition directly accounts for one tenth of nitrogen entering the Bay and could contribute closer to four tenths when considering indirect contributions from deposition to other land uses and freshwaters in the watershed (Alliance for the Chesapeake Bay, 1993).

2. Summary of Effects

Eutrophication effects include damages to fish stocks and changes in vegetation stocks. Decreased nitrogen oxide emissions are likely to result in larger eutrophication effects in ocean and estuarine ecosystems where deposition accounts for a large portion of the nitrogen budget.

C. Linkages to Economic Valuation

Attention is next awarded to evaluating the impacts on particular service flows provided by ocean and estuarine ecosystems, with the motivation being the economic valuation of changes in services flows. **Exhibit 2 [Summary of Impacted Service Flows]** displays the likely distribution of impacts from changes in nitrogen oxides emissions. Shaded cells indicate provision of a service by the ecosystem, and shaded cells marked with bolded **Xs** indicate susceptiblity to damage or change. Changes in the nitrogen levels within ocean and estuarine ecosystems will potentially result in eutrophication problems. Eutrophication affects the provision of service flows by these ecosystems by altering water quality and species composition. Recreational service flows such as birdwatching/wildlife viewing, scenic beauty, boating/water activities, and fishing might potentially be altered with such changes.

To assess the magnitude of the physical effects in economic terms, several linkages must be established: (1) linkage between nitrogen oxide emissions and nitrogen dynamics of the ecosystem; (2) linkage between nitrogen dynamics and fish stocks, plant life, and water quality; (3) linkage between fish stocks, plant life, and water quality and the quality of service flow provision; and (4) linkage between the demand for service flows and their quality.

The first two linkages address the physical response to changes in emissions, while the latter two linkages examine the human response and valuation of changes in physical conditions and service flow provision. Some research in areas such as the Chesapeake Bay has focused on the first two linkages. This research concentrates on deriving the contributions of deposition to the ecosystem's nitrogen budgets and water quality and then explores the repercussions of changes in water quality and nutrient levels on aquatic and terrestrial species.

The third and fourth linkages address behavioral responses to changes in ocean and estuarine systems. These changes can be linked to economic values of ocean and estuarine ecosystem service flows. The linkage of the provision of service flows to some species such as fish and factors such as water quality appear somewhat straight forward and appropriate for recreational service flows. Other types of indirect services might be more difficult to relate to these items. Appendix 3 includes one study that examined the effects of eutrophication on certain recreational service flows. Boating, beach use, and striped bass fishing service flows were studied by Bockstael et al. 1989 in the Chesapeake Bay Region. Average aggregate benefits (in $1987) from a twenty percent improvement in water quality calculated using the travel cost method were approximately $ 35 million for beach use, $5 million for boat use, and $1 million for striped bass fishing. Average aggregate benefits ($1987) estimated using the contingent valuation method were approximately $91 million. In terms of benefit transfer, this and other water quality studies on ocean and estuarine systems appear useful for recreational service flows. Other types of service flows might be more difficult to characterize using existing studies.

7. Deserts

A. Service Flows

Desert ecosystems located within the United States include the Great Basin, Great Salt Lake, Mojave, and Great American. Desert ecosystems are marked by the presence of high day time temperatures and water loss. Flora and fauna are determined by the level of water that is maintained in the ecosystem. Desert ecosystems provide several services to individuals. Direct recreational services include hiking, hunting, or camping opportunities and direct nonuse services offered include scenic beauty and waste sink services.

B. Physical Effects

Plant production in desert grasslands is greatly limited by water and typically coincides with infrequent and variable precipitation events, but direct relationships between precipitation and growth are difficult to specify (Fisher et al. 1988). In addition to water, nitrogen is noted as the next significant limiting factor to desert plant growth. It is likely that additional nitrogen deposition on desert ecosystems would lead to growth responses in arid vegetation even without water.

C. Linkages to Economic Valuation

We were unable to identify any economic studies evaluating service flows from desert ecosystems.

8. Grasslands

A. Service Flows

Grassland ecosystems are scattered throughout the United States. They are the dominant in the area from the western edge of eastern forests to the moderate altitudes of the Rocky Mountains. Other ecosystems are located in the northwest and the southwest. Grassland varieties include annual, bunchgrass steppe, northern mixed-grass prairie, shortgrass prairie, southern mixed-grass prairie, tallgrass prairie, and desert. Grasslands provide ‘limited service flows to individuals. Direct services associated with grassland ecosystems might include scenic beauty.

B. Physical Effects

The primary nitrogen contributors to unfertilized grassland ecosystems in the United States include NH_4, NO_3, and $R\text{-}NH_2$ via rain, snow, or dry deposition. Deposition is a more significant contributor in the northeastern Great Plains relative to grasslands located in the west. There is considerable spatial variation in nitrogen inputs to grassland ecosystems across the United States (Woodmansee 1978).

Nitrogen oxides can injure the flora of grasslands through nitrogen loading and acid deposition. Changes in community structure are also possible due to imposed changes in nitrogen fluxes. Ozone could potentially damage the grassland vegetation, alter the allocation and production of carbohydrates in the plants, or alter the grasslands floristic composition (Aldy, 1994).

C. Linkages to Economic Valuation

We were unable to identify any economic studies evaluating service flows from grasslands.

9. Summary

The previous sections offered ecosystem by ecosystem discussions of potential physical effects from nitrogen oxide controls, with some discussion of the linkages between these effects and the provision of service flows. This section devotes more emphasis to the understanding of these linkages and the information necessary to quantify and monetize effects in different ecosystems. The examination of linkages between physical processes and economic values is shaped by two influences: (1) the likelihood of physical effects from the Title IV controls and (2) the extent of existing scientific and economic research relevant to the linkage. Throughout the subsequent discussion, emphasis is awarded to both of these influences.

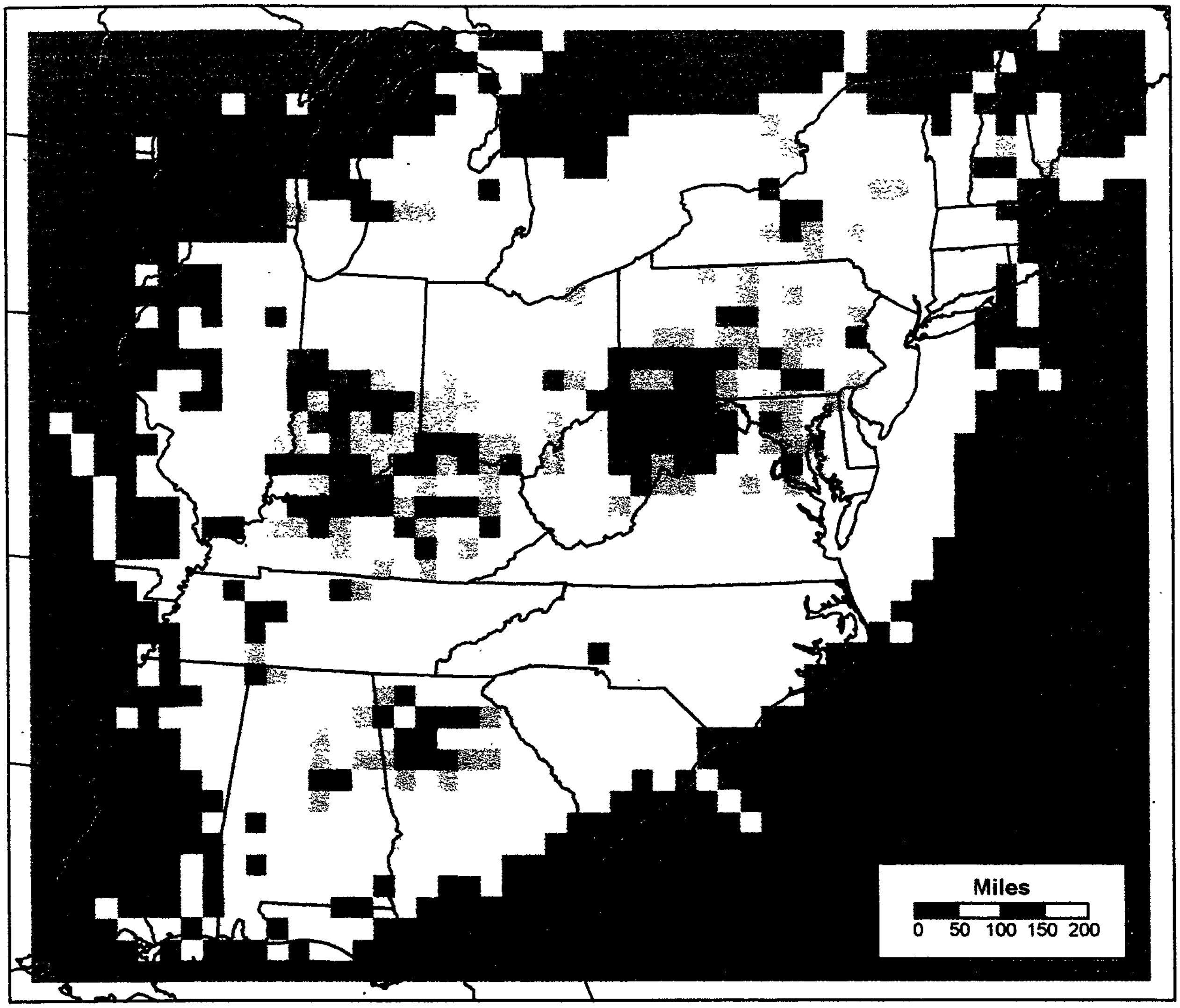
A. Emissions/Deposition Changes from the Title IV controls

The Title IV NO_x controls require reductions from specific types of coal-fired power plants. The regulation specifically targets plants operating dry bottom wall-fired or tangentially fired boilers. As noted previously, approximately 179 boilers at 110 facilities will be affected by the Phase I requirements. SAI modeled the expected changes in emissions of SO_2, SO_4, NO, NO_2, NH_3, PM, and O_3 for the Year 2000. Several emissions scenarios were modeled including: year 2000 base case emission estimates (E1); year 2000 with Title I controls on Title IV sources (E2); year 2000 with Title IV controls (E3); and year 2000 with an additional 75 percent NO_x emission controls on Title IV sources (E4). After looking at the emissions modeling results, the E4 scenario was selected as the background scenario for this discussion. This selection was made on the basis of the magnitude of the emissions changes under the different scenarios. The E4 scenario was chosen because it addresses the most stringent controls and thereto the greatest emissions reductions, providing greater capacity for applying the information organized as part of the inventory of physical effects.

The SAI emissions modeling work reveals that only a limited area is impacted by the regulation under consideration. The area affected includes ecosystems contained within New York, Pennsylvania, Ohio, Indiana, Illinois, Wisconsin, Michigan, West Virginia, Virginia, Kentucky, Tennessee, North Carolina, South Carolina, Mississippi, Alabama, and Georgia. Data are not currently available to explicitly characterize the spatial distribution of ecosystem categories within this area, but land use figures offer some sense of potential impacts. Total area estimates (km^2) by land use category for the affected region are as follows: urban (69,834); agricultural (1,065,672); rangeland (16); deciduous forest (643,814); coniferous forest (151,740); mixed forest (445,674); water (223,788); barren land (3); and non forest wetlands (15,469). Forest, agriculture, and water are the dominant land uses, followed thereafter by urban and non forested wetlands.

Exhibits 8, 9, and 10 present the SAI modeling predictions for total nitrogen, nitric acid, and ozone under scenario E4. The results are modeled using a grid framework. Data on emissions, deposition, pollutant levels, and land use composition are estimated for each grid cell. This modeling approach influences the aesthetic nature of the GIS exhibits. The color scheme is consistent across all three exhibits, with red tones illustrating greater changes and blue tones

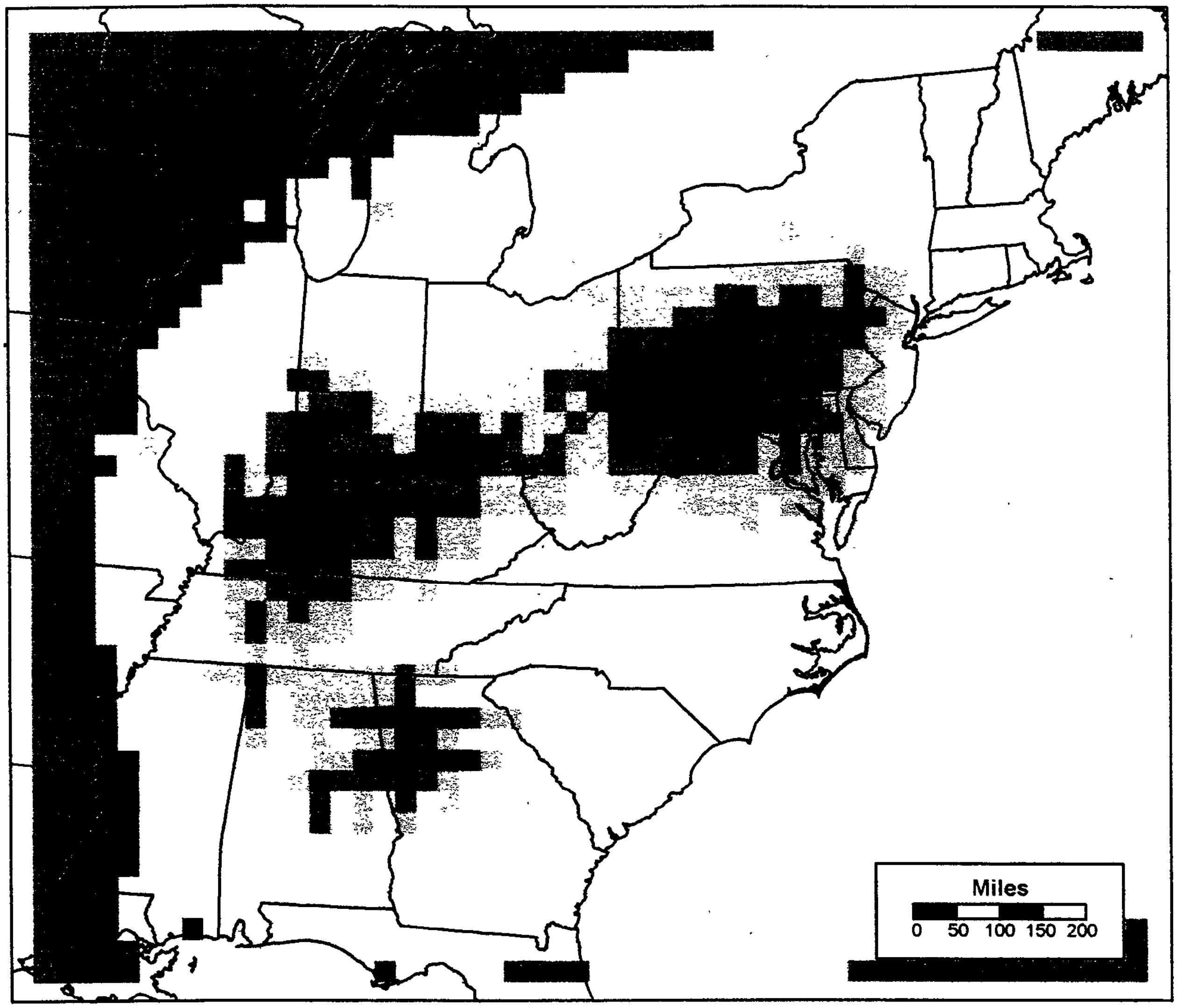
Exhibit 8
Change in the Annual Average Deposition of Total Nitrogen
Base Case (E1) - High NOx Control Case (E4) for the Year 2000



Reduction in Nitrogen Deposition (kg/ha)

.00 to .20
.20 to .40
.40 to .60
.60 to .80
.80 to 1.00
1.00 to 1.20
1.20 to 1.40
1.40 to 4.65

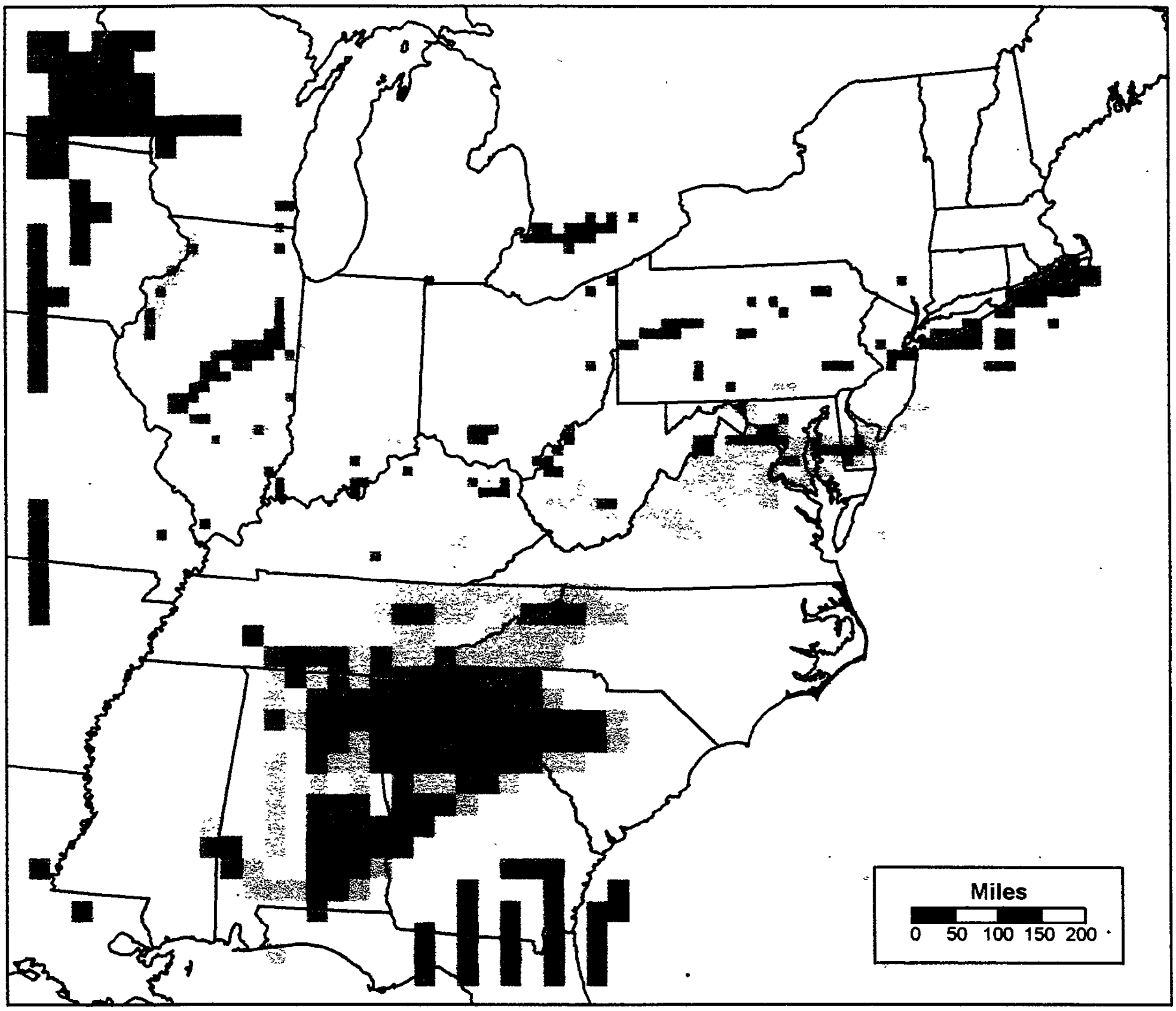
Exhibit 9
Change in the Annual Average Concentration of Nitric Acid
Base Case (E1) - High NOx Control Case (E4) for the Year 2000



Reduction in Nitric Acid Concentration (µg/m3)

.00	to	.10
.10	to	.20
.20	to	.30
.30	to	.40
.40	to	.50
.50	to	.60
.60	to	.70
.70	to	1.50

Exhibit 10
Change in Annual Average of Daily Maximum Ozone Concentrations
Base Case (E1) - High NOx Control Case (E4) for the Year 2000



Reduction in Ozone Concentration (ppb)

-.80	to	.00
.00	to	.40
.40	to	.80
.80	to	1.20
1.20	to	1.60
1.60	to	2.00
2.00	to	2.40
2.40	to	6.96

representing smaller changes relative to base line. Each exhibit displays the results for one pollutant. As such, the results are expressed in different units and the colors represent different magnitudes of change within Exhibits 8, 9, and 10. GIS presentation of the SAI modeling results facilitates the understanding and characterization of the emission and deposition changes from the Title IV controls. All three exhibits display the change between predicted baseline and E4 scenario results for the year 2000.

Exhibit 8 presents the change in annual average deposition (kg/ha) of total nitrogen for the year 2000. Under scenario E4 the largest reductions in total nitrogen relative to baseline range from approximately 1.40 kg/ha to 4.65 kg/ha. Spatially, these changes appear to be concentrated in northern Georgia, northern West Virginia, eastern Pennsylvania, southern Ohio, northern Kentucky, and southern Indiana. A large proportion of the affected area seems to experience only a modest change in annual average deposition of total nitrogen, ranging approximately from 0.20 to 0.60 kg/ha. Baseline deposition estimates range from 0.2 kg/ha to 18.7 kg/ha throughout the affected area, with much of the area having baseline deposition of approximately 4 kg/ha to 8 kg/ha. The modeling predictions intimate that the greatest changes occur in areas with baseline deposition of 8 kg/ha, and the changes represent approximately 18 to 60 percentage reductions in deposition from baseline.

Exhibit 9 displays the change in annual average concentration ($\mu g/m^3$) of nitric acid for the Year 2000. Spatially, the largest reductions in nitric acid concentration (0.70 to 1.50 $\mu g/m^3$) relative to baseline are found in areas similar to where the larger total nitrogen changes occur under scenario E4. This includes southern Pennsylvania, northern West Virginia, northern Kentucky, southern Indiana, southern Ohio, northern Georgia, and northern Alabama. Baseline annual average concentrations of nitric acid range from 0.7 $\mu g/m^3$ to 7.9 $\mu g/m^3$, with a mean concentration of approximately 3.1 $\mu g/m^3$. On average this suggests that the greatest changes in annual average concentration represent changes from baseline ranging from 23 to 48 percent. Much of the affected area appears to experience moderate changes of 0.10 $\mu g/m^3$ to 0.40 $\mu g/m^3$.

Exhibit 10 depicts the change in annual average of daily maximum ozone concentrations (ppb). The largest reductions in annual average daily maximum ozone concentrations (ppb) under scenario E4 range from 2.40 to 6.96 ppb. Spatially, these reductions are quite concentrated, falling primarily in northern Georgia, northern South Carolina, and eastern Alabama. Baseline daily maximum ozone concentrations range from 36.1 to 136.7 ppb. The largest reductions in daily maximum ozone concentration fall in areas with baseline daily maximum concentrations of 100 ppb. This suggests that the greatest reductions represent approximately 2.4 to 7.0 percentage changes relative to baseline. The majority of the affected area seems to experience modest reductions of approximately 0 ppb to 0.8 ppb.

Exhibits 11, 12, and 13 present land use acreage estimates by the different emissions/deposition/concentration change categories for ozone (ppb), total nitrogen (kg/ha), and nitric acid ($\mu g/m^3$). These data are helpful in understanding the type of land use acreage (and perhaps ecosystem) most affected by the Title IV NO_x controls. For example, the greatest reductions in ozone (> 2.4 ppb), total nitrogen (> 1.4 kg/ha), and nitric acid ($\mu g/m^3$) occur