October 2009

Analysis of Ecosystem Benefits for Reductions of NO_x and SO_x

Methodology Report

Prepared for

Amy Lamson
U.S. Environmental Protection Agency
Office of Air Quality Planning and Standards (OAQPS)
Air Benefit and Cost Group (ABCG)
(MD-C439-02)
Research Triangle Park, NC 27711

Prepared by

George Van Houtven
Paramita Sinha
Jennifer Phelan
Bill Cooter
Mary Barber
Michele Cutrofello
Jennifer Schimek
RTI International
Research Triangle Park, NC 27709

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ACRONYMS

Al	aluminum
ANC	acid neutralizing capacity
APS	Albemarle and Pamlico Sounds
CAA	Clean Air Act
CAFOs	Consolidated Animal Feeding Operations
CAIR	Clean Air Interstate Rule
CAL FIRE	California Department of Forestry and Fire Protection
CLE	critical load exceedance
CO_2	carbon dioxide
CPI-U	Consumer Price Index-All Urban Consumers
CSS	coastal sage scrub
CV	contingent valuation
DO	dissolved oxygen
DOI	Department of the Interior
EPA	Environmental Protection Agency
ESRI	Environmental Systems Research Institute, Inc.
FASOMGHG	Forest and Agricultural Sector Optimization Model—Green House Gas
FHWAR	Fishing, Hunting, and Wildlife Associated Recreation Survey
GBMM	Grid-Based Mercury Loading Model
GHG	greenhouse gas
GIS	geographical information system
HABs	harmful" algal blooms
HPF	household production function
MCF	mixed conifer forests
MEA	Millennium Ecosystem Assessment
MRFSS	Marine Recreation Fishing Statistics Survey
NASF	National Association of State Foresters
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NO_x	nitrogen oxides

OLS	ordinary least squares
RA	regulatory alternatives
RB	regulatory baseline
REA	risk and exposure assessment
RFF	Resources for the Future
RIA	regulatory impact analysis
RUM	random utility model
SAV	submerged aquatic vegetation
SERAFM	spreadsheet-based ecological risk assessment for the fate of mercury
SNAAQS	Secondary National Ambient Air Quality Standards
SO_x	sulfur oxides
SP	stated preference
SRB	S-reducing bacteria
TNP	total nitrogen and phosphorus
USGS	U.S. Geological Survey
VABRM	Virginia Blue Ridge Mountains
VHB	Viscusi, Huber, and Bell
WCS-MLM	watershed characterization system mercury loading model
WTP	willingness to pay

SECTION 1
INTRODUCTION

The U.S. Environmental Protection Agency (EPA) is conducting a joint review of the existing Secondary National Ambient Air Quality Standards (SNAAQS) for nitrogen oxides (NO_x) and sulfur oxides (SO_x). As part of this review process, EPA completed a risk and exposure assessment (REA) that evaluated the exposures of ecological receptors to both ambient and deposited species of NO_x and SO_x, as well as their transformation products, and assessed the risks associated with these exposures (EPA, 2009). The REA also included an assessment of ecosystem services, which examined how ecological exposures and risks affect the well-being that humans derive from ecosystems.

As part of the federal rulemaking process, a regulatory impact analysis (RIA) is required to evaluate the costs and benefits of proposed regulations that have significant economic effects. In contrast to the REA, which focused on the overall ecological impacts and losses in ecosystem services due to past and current NO_x/SO_x levels, the RIA benefits analysis is focused on identifying, quantifying, and valuing future reductions in NO_x/SO_x levels as a result of specific regulatory alternatives and relative to an assumed future regulatory baseline. The main purpose of this report is to describe a methodological framework for assessing the ecosystem benefits of regulations to reduce ambient NO_x and SO_x levels. However, this framework should also be applicable for assessing the ecosystem benefits of other measures that reduce future nitrogen (N) or sulfur (S) deposition in the United States (e.g., future revisions to the particulate matter NAAQS, Clean Air Interstate Rule [CAIR] replacement rule).

Figure 1-1 illustrates the general conceptual framework guiding the methods development for this report. To properly assess the benefits of NO_x/SO_x controls, conditions resulting from the proposed regulatory alternatives must be evaluated relative to appropriately defined reference conditions. That is, what would NO_x/SO_x levels be without these proposed rules? As shown in Figure 1-1, the reference condition is specified as the "regulatory baseline." In particular, it accounts for expected emission controls associated with promulgated federal regulations that are anticipated to be implemented prior to the analysis year, but no additional measures to reduce N+S deposition. As explained and quantified in the REA, N+S deposition are associated with a range of adverse effects on aquatic and terrestrial ecosystems, and these effects reduce the flow of services from ecosystems to humans. The red arrows in Figure 1-1 trace the effects of NO_x-SO_x levels on ecosystem services under the regulatory baseline conditions, and the green arrows do the same for the regulatory alternatives. The ecosystem effect and ecological indicator boxes in the diagram identify some of the main

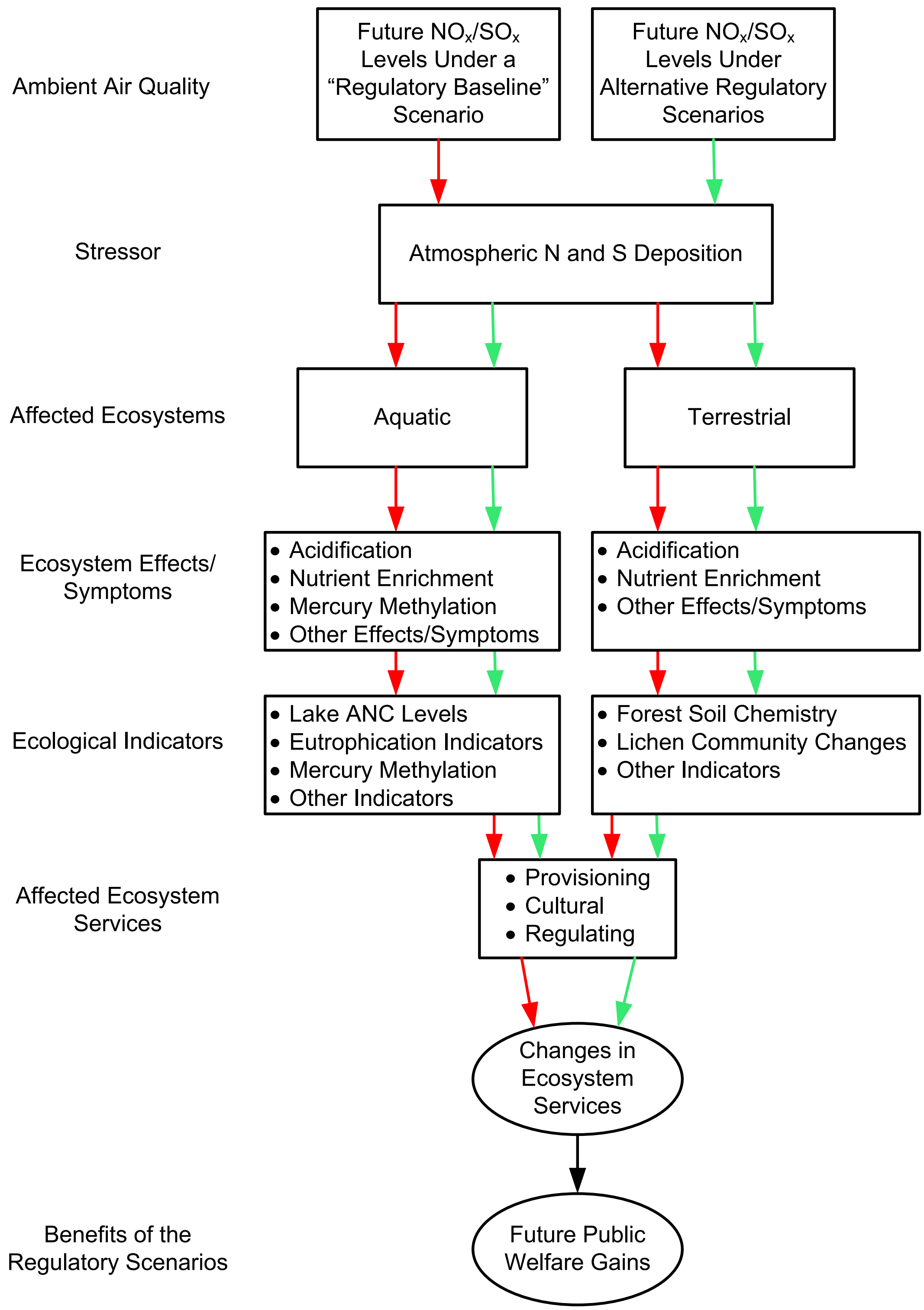

Figure 1-1. Conceptual Framework for Linking Changes in Ambient NO_x and SO_x Levels to Changes in Ecosystem Services and Benefits

categories and examples that are addressed in the REA and also in this report. Measures that lower ambient NO_x and SO_x concentrations are expected to result in lower negative effects and thus higher flows of ecosystem services. The benefits of the regulatory alternatives are derived from the *difference* (i.e., change) in ecosystem service flows between the regulatory baseline and regulatory alternative scenarios. In other words, the benefits are the enhancements in human well-being that result from the increase in ecosystem services.

Figure 1-2 provides additional detail regarding the temporal dimensions of the benefits assessment methodology. The top panel displays a simplified and general representation of the expected paths of N+S deposition under the alternative scenarios. First, the regulatory baseline scenario is represented by a downward-sloping line from the current period into the future.[1] Second, beginning in the year when a revised standard is implemented, it is expected that NO_x/SO_x levels would decline more rapidly (at least initially) than under the regulatory baseline. Two alternative proposed standards are represented in Figure 1-2. For the purposes of the RIA, a future year (2020) is selected as the main point of analysis. National air quality modeling runs for that year will be conducted to estimate and compare annual deposition under the regulatory baseline (point B) and the regulatory alternatives (C and D).

The bottom panel of Figure 1-2 represents the assumed time paths of N+S deposition that will be used for the benefits assessment methods described in this report. These time paths represent a further simplification of the patterns shown in the top panel; however, importantly they include the same expected deposition levels in 2020 (points B, C, and D). To make the initial analysis more tractable, *all* future declines in deposition from current levels are assumed to occur in 2020. Before 2020, it is assumed that there is no difference in deposition between the baseline and alternative scenarios, and after 2020, the differences are the same in each year. These assumptions can eventually be changed to incorporate more complex and realistic time paths, but they provide a useful starting point for defining the benefits analysis methods.

Following the framework developed for the REA and ecosystem services analysis, this report includes five main sections (after this one), each addressing one of following main ecological effect categories:

- aquatic acidification
- terrestrial acidification

[1] Although the general trend nationally is expected to be downward sloping, there may be some specific (isolated) locations where the regulatory baseline is upward sloping.

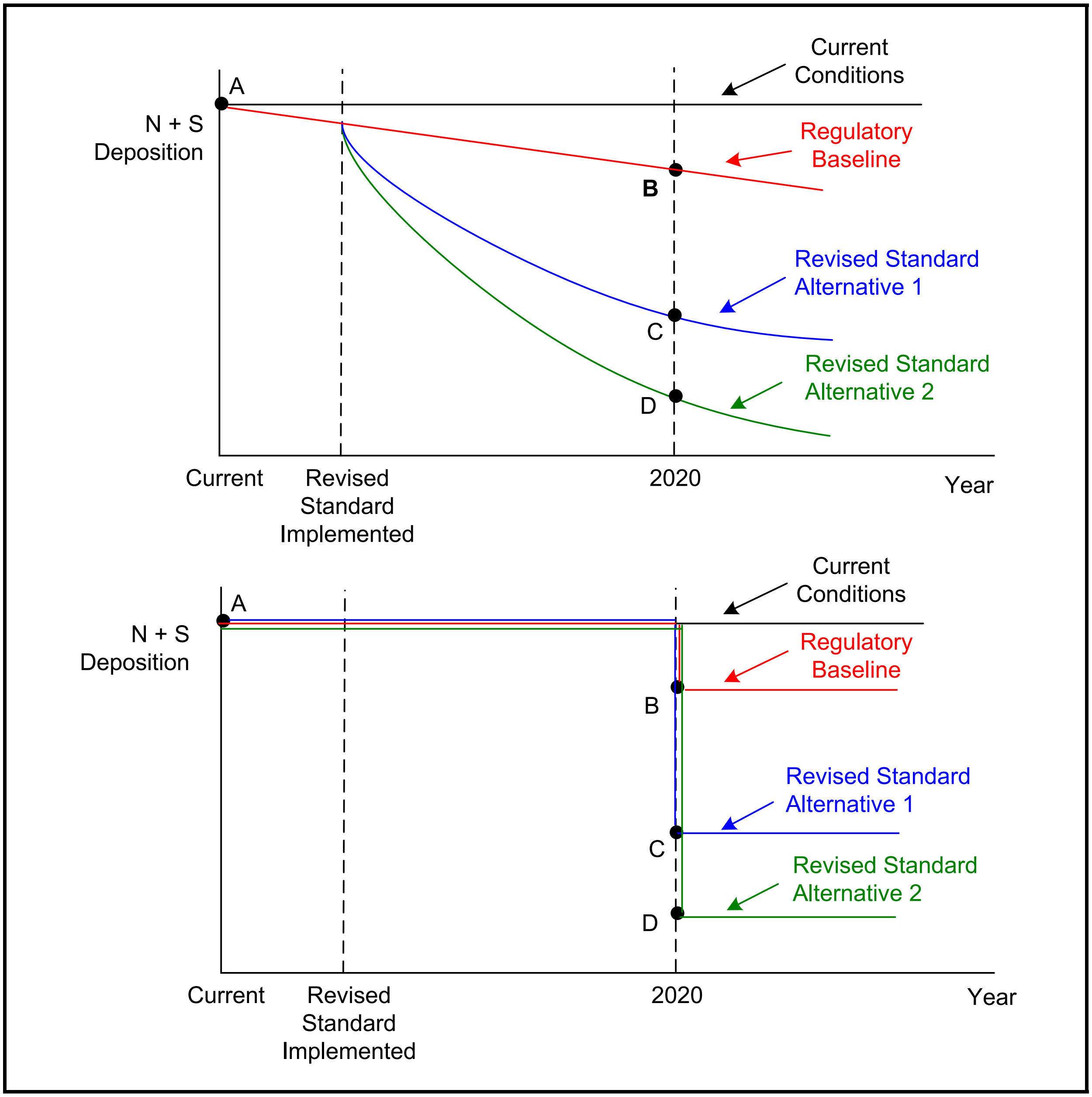

Figure 1-2. Temporal Framework: Representation of Future Time Paths for N+S Deposition under Alternative Regulatory Scenarios

- aquatic nutrient enrichment
- terrestrial nutrient enrichment
- mercury methylation

Following the recommendations of EPA's Science Advisory Board Committee on Valuing the Protection of Ecological Systems and Services (EPA, 2009), each section begins by "developing a conceptual model of the relevant ecosystem and the ecosystem services that it generates." These models highlight the main linkages between N+S deposition, ecological symptoms and endpoints, and adversely affected ecosystem services. As such, the models provide a roadmap for identifying and, to the extent possible, quantifying and monetizing the main ecological benefits of reduced NO_x/SO_x levels.

For each of the five main ecological effect categories, Table 1-1 identifies the main ecosystem services that are adversely affected by N+S deposition in the United States. It also lists the specific areas where quantitative methods for linking changes in deposition to monetary benefits have been developed. These affected services and quantitative methods are described in each of the following sections.

Table 1-1. Identification and Quantification/Monetization of Main Affected Ecosystem Services

Affected Ecosystem Services	Quantitative Benefits Assessment Methodology
Aquatic Acidification (Section 2)	
Provisioning Services	
Commercial fishing	
Cultural Services	
Total	▪ NY Adirondack lakes ▪ VA Blue Ridge streams
Recreational fishing	▪ NY Adirondack lakes
Regulating Services	
Biological control	
Terrestrial Acidification (Section 3)	
Provisioning Services	
Commercial forest products	▪ Sugar maple trees in NE U.S. ▪ Red spruce trees in NE U.S.
Cultural Services	
Recreation and aesthetic	
Nonuse services	
Regulating Services	
Erosion control	
Water regulation	
Climate regulation	

(continued)

Table 1-1. Identification and Quantification/Monetization of Main Affected Ecosystem Services (continued)

Affected Ecosystem Services	Quantitative Benefits Assessment Methodology
Aquatic Enrichment (Section 4)	
Provisioning Services	
Commercial fishing	▪ Crab fishing in NC Neuse estuary
Cultural Services	
Recreational	▪ Fishing in Chesapeake Bay[a] and NC Albemarle-Pamlico Sound ▪ Boating in Chesapeake Bay[a] ▪ Beach use in Chesapeake Bay[a]
Aesthetic	▪ Near shore Chesapeake Bay residents[a]
Nonuse services	▪ Chesapeake Bay[a]
Regulating Services	
Erosion control	
Storm protection	
Terrestrial Enrichment (Section 5)	
Provisioning Services	
Livestock production	
Forest products	
Cultural Services	
Recreation	
Aesthetics	
Nonuse Services	
Regulating Services	
Erosion Control	
Fire regulation	
Biological Control	
Water regulation	
Mercury Methylation (Section 6)	
Provisioning Services	
Commercial and subsistence fishing	
Cultural Services	
Recreation	
Nonuse Services	
Regulating Services	
Biological Control	

[a] Gap exists for quantitative link between nutrient loadings to estuary and estuarine water quality.

Section 2 focuses on aquatic acidification and provides an overview of the main ecosystem services affected by acidification of freshwater. To demonstrate methods for assessing the benefits of reduction in aquatic acidification, this section applies and adapts results from two of the REA case studies. First, it applies the case study of Adirondack lakes to develop two alternative methods for assessing the benefits of improvements in recreational fishing services and cultural services in general, as a result of reductions in lake acidification in this part of the country. Second, it uses the Shenandoah case study results to develop a method for assessing the benefits of reducing the percentage of streams impaired by acidification in the Blue Ridge region of Virginia.

Section 3 focuses on terrestrial acidification. In addition to describing the linkages between N+S deposition and impairments to forest-based ecosystem services, it describes a method for assessing the benefits of improved commercial forest productivity. Building on the methods described in the REA, this part of the analysis focuses on the benefits derived from markets for sugar maple and red spruce wood products. This section also proposes methods for extending this analytical approach to include additional data and additional tree species.

Section 4 focuses on aquatic nutrient enrichment due to nitrogen deposition, which is particularly a problem for estuaries in the Mid-Atlantic and Northeastern regions of the United States. In addition to describing a general conceptual framework, it focuses on two main estuarine systems to describe benefits assessment methods. First, it describes methods for quantifying recreational, aesthetic, and nonuse benefits associated with reductions in eutrophication in the Chesapeake Bay. One of the main limitations for applying these methods is that a key modeling gap exists for linking changes in N loadings to the Bay to changes in water quality in the Bay. Second, it describes methods for estimating fishing benefits in North Carolina estuaries.

Section 5 focuses on terrestrial nutrient enrichment. Although these enrichment effects potentially affect large areas and many terrestrial systems in the United States, methods for quantifying the relevant cause-and-effect relationships are currently not available. Therefore, this section provides a more qualitative discussion of the potentially affected systems, services and benefits. The discussion focuses on three main terrestrial ecosystems in the United States– California coastal sage scrub (CSS), California mixed conifer forests (MCF), and western grasslands.

Section 6 focuses on mercury methylation. Like the previous section, it provides a more qualitative discussion of the potentially affected systems, services, and benefits. Currently available data and models are not adequately developed to provide quantitative methods for predicting the mercury-related benefits of reductions in S deposition.

1.1 Reference

U.S. Environmental Protection Agency (EPA). 2009. *Risk and Exposure Assessment for Review of the Secondary National Ambient Air Quality Standards for Oxides of Nitrogen and Oxides of Sulfur.* Research Triangle Park, NC: U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards. EPA-452/R-09-008a.

SECTION 2
AQUATIC ACIDIFICATION

2.1 Overview of Adversely Affected Ecosystems and Services

Atmospheric deposition in the United States is often a major cause of surface water acidification, which harms fish and wildlife and reduces the human welfare that is derived from these natural resources (EPA, 2008b). Based on factors including surficial geology and soil type, which make some areas more vulnerable to the effects of acidifying deposition, Figure 2-1 shows areas in the United States that are potentially most sensitive to aquatic acidification.

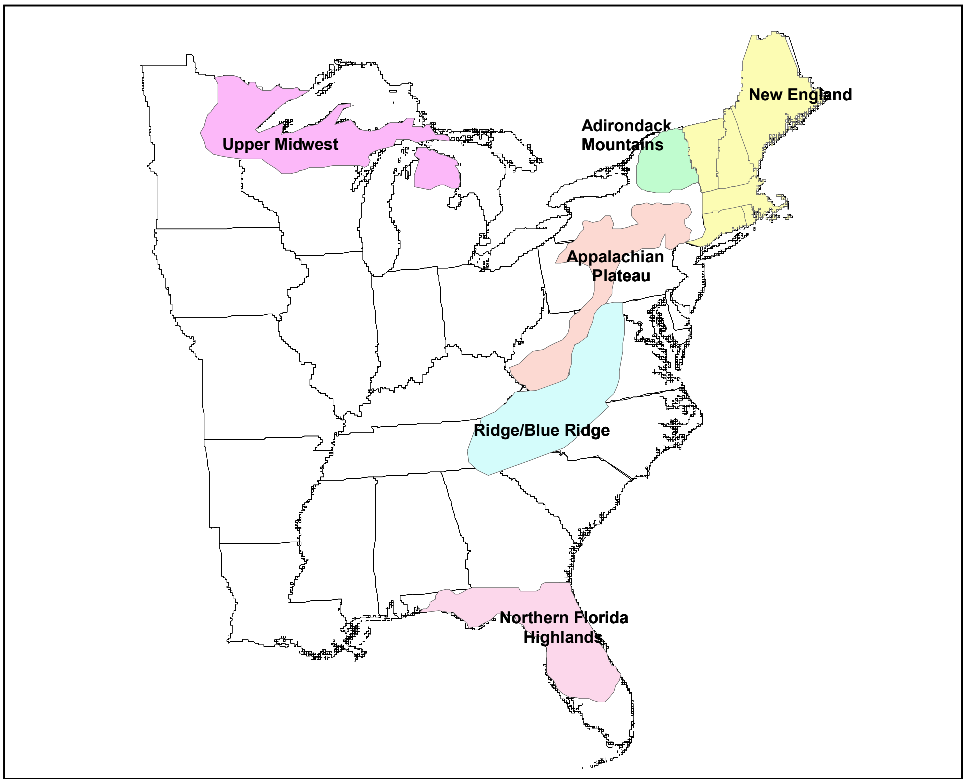

Figure 2-1. Ecosystems Sensitive to Acidifying Deposition in the Eastern United States (EPA, 2009)

Figure 2-2 displays a conceptual model, which highlights the main processes and adverse outcomes associated with aquatic acidification. High levels of N+S deposition, particularly in areas with soils containing relatively low levels of alkaline chemical bases such as calcium or magnesium ions, often lead to acidification of surface waters such as lakes and streams. Two of the main indicators of acidity are surface water pH and acid neutralizing capacity (ANC). High

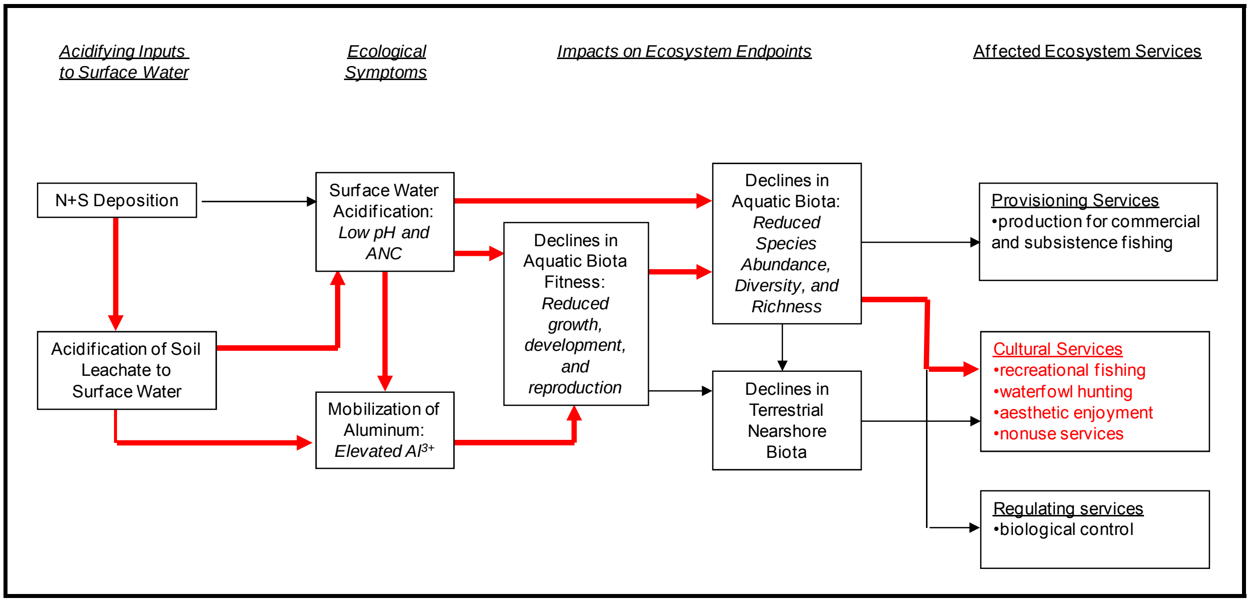

Figure 2-2. Conceptual Diagram of Ecosystem Service Impairments Associated with Aquatic Acidification[a]

[a] Red arrows and fonts highlight the areas for which quantitative models are described in this section. Bold arrows represent the stronger and better established cause-and-effect relationships.

levels of acidity can in turn mobilize and increase concentrations of aluminum (Al^{3+}), which are toxic to fish.

In general, moderate shifts in ANC levels may result in changes in species composition, where acid-sensitive species are replaced by less sensitive species. At more extreme acidification levels, however, species richness, defined as the total number of species occupying a system, may be affected. Research has shown that the number of fish species present is positively correlated with ANC (Driscoll et al., 2003). As summarized in Table 2-1, several ANC thresholds have been observed at which lakes and fish are affected. For example, recent research in the Adirondacks region of New York indicates that aquatic biota begin to exhibit effects at an ANC of 50 microequivalents per liter (μeq/L) (Chen and Driscoll, 2004).

Evidence of both chronic and episodic acidification of surface waters is particularly evident in the eastern and northeastern United States, where levels of N+S deposition have also been relatively high in recent decades. These surface waters support a wide variety of ecosystem services, many of which can be affected adversely by acidification. Because acidification primarily affects the diversity and abundance of aquatic biota, it also primarily affects the ecosystem services that are derived from the fish and other aquatic life found in these surface waters.

Table 2-1. Aquatic Status Categories Associated with Ranges of ANC Levels

Category Label ANC Levels[a] and Expected Ecological Effects		
Acute concern	<0 micro equivalent per liter (μeq/L)	Complete loss of fish populations is expected. Planktonic communities have extremely low diversity and are dominated by acidophilic forms. The number of individuals in plankton species that are present is greatly reduced.
Severe Concern	0–20 μeq/L	Highly sensitive to episodic acidification. During episodes of high acidifying deposition, brook trout populations may experience lethal effects. Diversity and distribution of zooplankton communities decline sharply.
Elevated concern	20–50 μeq/L	Fish species richness is greatly reduced (i.e., more than half of expected species can be missing). On average, brook trout populations experience sublethal effects, including loss of health, reproduction capacity, and fitness. Diversity and distribution of zooplankton communities decline.
Moderate Concern	50–100 μeq/L	Fish species richness begins to decline (i.e., sensitive species are lost from lakes). Brook trout populations are sensitive and variable, with possible sublethal effects. Diversity and distribution of zooplankton communities also begin to decline as species that are sensitive to acidifying deposition are affected.
Low concern	>100 μeq/L	Fish species richness may be unaffected. Reproducing brook trout populations are expected where habitat is suitable. Zooplankton communities are unaffected and exhibit expected diversity and distribution.

[a] Source: EPA (2009)

2.1.1 Effects on Provisioning Services

Food and freshwater are generally the most important provisioning services provided by inland surface waters (Millennium Ecosystem Assessment [MEA], 2005). Whereas acidification is unlikely to have serious adverse effects on, for example, water supplies for municipal, industrial, or agricultural uses, it can limit the productivity of surface waters as a source of food (i.e., fish). In the northeastern United States, the surface waters affected by acidification are not a major source of commercially raised or caught fish; however, they are a source of food for some recreational and subsistence fishers and for other consumers. Although data and models are available for examining the effects on recreational fishing (see Section 2.1.2), relatively little data are available for measuring the effects on subsistence and other consumers. For example, although there is evidence that certain population subgroups in the northeastern United States, such as the Hmong and Chippewa ethnic groups, have particularly high rates of self-caught fish consumption (Hutchison and Kraft, 1994; Peterson et al., 1994), it is not known if and how their consumption patterns are affected by the reductions in available fish populations caused by surface water acidification.

2.1.2 Effects on Cultural Services

Inland surface waters support several cultural services, such as aesthetic and educational services; however, the type of service that is likely to be most widely and significantly affected by aquatic acidification is recreational fishing, since it depends directly on the health and abundance of aquatic wildlife. Other recreational activities such as hunting and birdwatching are also likely to be affected, to the extent that fish-eating birds and other wildlife are harmed by the absence of fish in acidic surface waters.

Recreational fishing in lakes and streams is among the most popular outdoor recreational activities in the northeastern United States. Data from the 2006 National Survey of Fishing, Hunting, and Wildlife Associated Recreation (FHWAR) indicate that more than 9% of adults in this part of the country participate annually in freshwater (excluding Great Lakes) fishing. The total number of freshwater fishing days occurring in those states (by both residents and nonresidents) in 2006 was 140.8 million days. Roughly two-thirds of these fishing days were at ponds, lakes, or reservoirs in these states, and the remaining one-third were at rivers or streams. Based on studies conducted in the northeastern United States, Kaval and Loomis (2003) estimated an average consumer surplus value per day of $35.91 for recreational fishing (in 2007 dollars). Therefore, the implied total annual value of freshwater fishing in the northeastern United States was $5.06 billion in 2006.

2.1.3 Effects on Regulating Services

In general, inland surface waters such as lakes, rivers, and streams provide a number of regulating services, such as hydrological regime regulation and climate regulation. There is little evidence that acidification of freshwaters in the northeastern United States has significantly degraded these specific services; however, freshwater ecosystems also provide biological control services by providing environments that sustain delicate aquatic food chains. The toxic effects of acidification on fish and other aquatic life impair these services by disrupting the trophic structure of surface waters (Driscoll et al., 2001). Although it is difficult to quantify these services and how they are affected by acidification, it is worth noting that some of these services may be captured through measures of provisioning and cultural services. For example, these biological control services may serve as “intermediate” inputs that support the production of “final” recreational fishing and other cultural services.

2.2 Methodology for Assessing the Benefits of Reductions in Aquatic Acidification

Quantifying the benefits derived from future reductions in aquatic acidification damages first requires a model linking NO_x/SO_x concentrations and deposition to surface water impairments. The REA describes the structure and results of such a model—the MAGIC model—which has been applied to two case study areas in the United States: the Adirondack State Park in New York and the Shenandoah region of Virginia. In both cases, the MAGIC model was used to predict impairments, measured in terms of ANC levels, for a selected sample of surface waters under both historical and projected future deposition conditions.

In this section, the Adirondack and Shenandoah case studies are used to demonstrate how MAGIC modeling results can be used to assess the benefits of reduced aquatic acidification in specific areas. These benefit assessment methods can be adapted and expanded to accommodate additional MAGIC results, as they become available. In particular, efforts are currently underway at EPA to apply MAGIC to waterbodies in the New England and Upper Midwest regions (see Figure 2-1) and to expand the number of waterbodies being modeled in the two original case study areas.[1]

Using the MAGIC framework for benefits assessment will require simulation runs for the modeled waterbodies that represent conditions under the regulatory baseline and the regulatory alternative scenarios. In other words, it will require 2020 CMAQ modeling results for the regulatory scenarios and assumptions about the corresponding time paths of deposition before and after 2020 (as shown in Figure 1-2). Assuming that these conditions are met, the next mains steps are to (1) extrapolate the modeled scenario estimates to a larger and more complete set of waterbodies in a defined area and (2) estimate the aggregate value of changes from the regulatory baseline to the regulatory alternative scenarios for a defined population of interest. The following two subsections outline specific approaches for addressing these steps, first using the Adirondacks case study framework and second using the Shenandoah framework.

2.2.1 Benefits of Reduced Acidification in Adirondack Lakes

The Adirondacks case study analysis focused on 44 lakes and estimated ANC levels at each lake under the alternative scenarios shown in Table 2-2. Using the MAGIC model, it predicted median ANC levels for the years 2005, 2020, 2050, and 2100 under "business-as-usual" conditions (i.e., accounting for expected emission controls associated with Title IV regulations but no additional measures to reduce N+S deposition). In contrast, the model run for

[1]Personal communication with Jason Lynch (EPA, Clean Air Markets Division) on 10-13-2009.

Table 2-2. Example of MAGIC Model Output for Lakes in the Adirondacks—Predicted Median ANC Levels (in µeq/L)

	MAGIC Model Simulations				
Year	**2005**	**2020[a]**	**2050[a]**	**2100[a]**	**1860[b] ("Background")**
Lake Name					
Clear Pond (61)	233.0	243.2	246.7	247.6	290.3
Long Pond (65)	73.5	78.3	80.4	81.2	106.4
Hope Pond	72.9	78.4	81.1	82.8	126.5
Second Pond	75.8	77.0	75.3	72.5	121.5
Squaw Lake	25.6	27.1	24.9	21.3	73.8
Indian Lake	1.4	6.2	6.2	5.1	52.2
Big Alderbed	67.5	72.6	74.5	75.6	124.1
Long Lake	−20.8	−15.4	−16.0	−17.6	34.4
Gull Pond	166.8	170.7	173.0	174.6	208.8
Little Lilly Pond	54.4	57.9	58.5	58.6	95.5
Upper Sister Lake	37.4	39.9	39.4	38.0	80.3
Dry Channel Pond	31.7	34.3	33.2	31.6	78.6
Bennett Lake	37.5	39.2	37.8	35.0	69.7
Effley Falls Pond	59.8	64.2	64.2	63.7	132.4
Parmeter Pond	85.7	91.7	94.3	95.4	134.8
North Lake	6.9	10.9	10.0	8.1	66.0
Razorback Pond	39.6	42.4	40.5	37.4	94.3
Snake Pond	12.3	15.5	14.5	13.4	78.5
South Lake	0.1	3.7	2.3	−0.2	56.6
Boottree Pond	59.0	63.2	65.3	66.1	84.5
Horseshoe Pond	63.0	70.0	73.4	74.8	117.6
Rock Pond	95.1	98.8	99.5	99.5	151.5
Antediluvian Pond	70.1	72.0	71.4	69.9	95.3
Seven Sisters Pond	−9.1	−6.9	−7.2	−8.1	21.9
Canada Lake	69.4	77.6	80.1	81.4	151.2
Bickford Pond	33.6	41.3	45.2	46.9	101.3
Wolf Pond	4.8	9.8	11.2	11.9	58.3
Blue Mountain Lake	126.7	129.3	127.8	125.2	184.3
Carry Falls Reservoir	133.2	140.8	144.1	145.8	205.8

(continued)

Table 2-2. Example of MAGIC Model Output for Lakes in the Adirondacks—Predicted Median ANC Levels (in µeq/L) (continued)

	MAGIC Model Simulations				
Year:	2005	2020[a]	2050[a]	2100[a]	1860[b] ("Background")
Lake Name					
Rocky Lake	58.6	66.3	68.3	68.8	113.7
Bog Pond	107.0	117.2	120.5	121.6	178.1
Clear Pond (82)	97.1	104.1	107.4	108.2	145.6
Seventh Lake	217.3	223.4	227.1	229.1	317.6
Trout Pond	53.4	61.9	65.7	67.6	127.2
Hitchins Pond	162.7	170.0	172.6	173.8	214.7
Piseco Lake	114.7	123.7	127.2	128.6	186.2
Mccuen Pond	46.0	50.2	51.7	52.4	90.0
Arbutus Pond	101.6	108.6	111.3	113.1	187.1
Witchhopple Lake	35.7	39.4	38.9	37.6	91.7
Willys Lake	−38.8	−33.5	−33.3	−33.4	47.5
Lower Beech Ridge Pond	−10.8	−6.9	−7.4	−8.8	41.5
Dismal Pond	−12.0	−7.6	−7.3	−7.6	40.4
Payne Lake	56.2	58.1	59.0	59.4	75.1
Whitney Lake	30.7	33.7	32.9	31.5	84.3

[a] Based on predicted future scenarios for N+S deposition, accounting for Title IV emissions controls.
[b] Represents background levels and levels that would eventually result from a "zero-out" of anthropogenic sources of N+S deposition.

the year 1860 represents ANC levels for "background" conditions by simulating the effect of zeroing out anthropogenic sources of N+S. For future benefits analyses, it is assumed that similar simulations for 2005 to 2100 will be developed to represent the regulatory baseline and regulatory alternative scenarios.

Below, two approaches are described for applying these case study results to assess the benefits of reduced lake acidification. Section 2.2.1.1 outlines a methodology for specifically assessing the recreational fishing benefits of reduced acidification in Adirondack and other New York lakes. Section 2.2.1.2 describes a broader ecosystem services approach for estimating the overall ecological benefits from reduced acidification of Adirondack lakes.

2.2.1.1 Recreational Fishing Benefits from Reduced Acidification in Adirondack and Other New York Lakes

The approach described in this subsection has been previously used to analyze the ecological benefits of the 1990 Clean Air Act (CAA) Amendments (Second Section 812 Prospective Project Team, 2007) and the economic impacts of the Clean Air Interstate Rule (CAIR) (IEc, 2008). To estimate the recreational fishing benefits, this approach applies an existing random utility model (RUM) that relates changes in lake acidity to a change in recreational fishing behavior throughout the study area. Applying the RUM model results requires that the MAGIC model results be extrapolated to the larger set of regional lakes included in the original RUM model estimation. This extrapolation can be accomplished using a random effects model, which relates modeled acidification levels to lake characteristics and geographic location.

<u>Random Effects Model for Spatial Extrapolation</u>

The random effects model is a multivariate regression model that uses data on lake characteristics to explain variation in modeled ANC levels. A random effects modeling approach using the MAGIC results from Table 2-2 is described in detail in Appendix 8 in EPA (2009). In that application, variables describing elevation, total area, and shoreline length were included as explanatory variables to capture physical differences between lakes. Binary variables indicating the counties in which the lakes are located were also included as proxies for a host of other location-specific factors for which data were not available, including subsurface geology and degree of forest cover. In addition, an annual time trend variable (T) was included to capture changes through time manifested in the greater system and not a specific lake.

Table 2-3 reports the results of the random effects model application used in the REA. Although the coefficients in this model are not statistically significant, the variables do lend some explanatory power to the model.[2] For future benefits analyses, the same model could be reestimated using new MAGIC results for the 35 included lakes. Alternatively, if resources permit, the model could be reestimated using MAGIC results for additional lakes and with additional lake characteristics data.

[2]To estimate the random effects model, it was first necessary to compare the subset of lakes considered in the ecological model (see Table 2-1) with the subset of lakes included in the database of lake characteristics contained within the RUM study. Nine of the 44 lakes were not usable for the analysis because they did not appear in the database of lake characteristics within the RUM. Those lakes excluded include the following: Bickford Pond, Bog Pond, Hope Pond, Little Lilly Pond, Lower Beech Ridge, Razorback Pond, Seven Sisters Pond, Snake Pond, and Witchhopple Lake.As a result, the analysis relied on data for a subset of 35 Adirondack lakes.

Table 2-3. Random Effects Model Results

Variable	Coefficient	Std. Error
Constant	−106.171	75.050
Elevation	−0.047	0.128
Area	0.125	0.074
ln(shoreline)	−36.005	18.802
T	0.108	0.013
Hamilton	9.430	27.760
Essex	55.149	46.894
Fulton	−16.793	80.273
Franklin	49.538	39.176
Herkimer	−38.655	40.142
Lewis	−19.160	45.899
Warren	24.924	66.423

The primary objective in developing the random effects model is to spatially extrapolate findings from the MAGIC modeled lakes to other lakes included in the original RUM study. The original study included a total of 2,586 lakes across New York State. The 35 lakes included in the random effects regression reported in Table 2-3 are located in Hamilton, Essex, Fulton, Franklin, Herkimer, Lewis, Warren, and St. Lawrence counties. Their explanatory value for lakes outside of this eight-county region is less certain. Therefore, a "tiered" extrapolation can be used, where the random effects model results are first extrapolated only to lakes in the Adirondack region represented by the modeled lakes and then extrapolated to the full suite of New York State lakes.

For the first tier (for the Adirondack region), the analysis is limited by two dimensions: (1) only including lakes within the eight counties containing the 35 modeled lakes and (2) limiting the analysis to lakes within the size range of the modeled lakes. This extrapolation excludes lakes in three counties within the Adirondack region—Clinton, Saratoga, and Oneida counties—because none of the 35 modeled lakes are in these counties. This assumption may lead to an understatement of the total benefits associated with decreased lake acidification in the Adirondack region, but it avoids some uncertainty associated with extrapolating ANC outside of the scope of the modeled region.

The second tier of the analysis (for all of New York State except New York City) is also limited to considering only lakes within the size range of the modeled lakes; however, it expands the extrapolation to include lakes outside of the eight-county geographic scope. To predict ANC values for lakes outside of the eight county region, an average of the eight-county binary variable coefficients from the random effects regression can be used. Further, as with the first tier of the analysis, all lakes with an area greater than the largest lake in the ecological subset of 35 can be "hardwired" to be unimpaired, because changes in their ANC levels are unlikely to be represented by the subset of modeled lakes A total of 62 lake sites were determined to be too large to be represented by the sample MAGIC data and were, therefore, hardwired.[3]

In addition to spatially extrapolating the MAGIC model results, a random effects modeling approach can be used to predict ANC values in future years and under alternative regulatory scenarios. For example, the time trend variable, T, in Table 2-3 captures the average predicted annual increment in ANC under the business as usual (BAU) baseline scenario. Therefore, this result was applied to predict future ANC levels for all extrapolated lakes under the BAU scenario. With MAGIC model predictions of ANC levels under alternative regulatory conditions, alternative time trend variables can also be included in the random effects regression. The new time trend coefficient estimates can then be used to predict future ANC levels for all extrapolated lakes under the alternative regulatory scenarios.

Application of ANC Thresholds

The economic model applied in this method is a repeated discrete choice RUM that describes lake fishing behavior of New York residents (Montgomery and Needelman, 1997). In particular, the model characterizes decisions regarding (1) the number of lake fishing trips to take each season and (2) the specific lake sites to visit on each fishing trip. One of the key explanatory variables in the lake choice decision model is a binary indicator of whether lakes are fishable or nonfishable.

To incorporate the MAGIC and random effects model predictions into this RUM framework, it is necessary to define a mapping between the ANC measures used in the former and the fishability indicator used in the latter. Previous applications linking these models have employed three ANC threshold assumptions—20 μeq/L, 50 μeq/L, and 100 μeq/L—to indicate

[3]Hardwired lakes (in order of decreasing size) include Lake Ontario, Lake Erie, Great Sacandaga Lake, Oneida Lake, Seneca Lake, Lake Champlain, Cayuga Lake, Lake George, Canandaigua Lake, Ashokan Reservoir, Cranberry Lake, Owasco Lake, Chautauqua Lake, Tupper Lake, Stillwater Reservoir, Keuka Lake, Pepacton Reservoir, Allegheny Reservoir, Raquette Lake, Cannonsville Reservoir, Indian Lake, Skaneateles Lake, Black Lake, Long Lake, Otsego Lake, Saratoga Lake, Mount Morris Reservoir, Salmon River Reservoir, Great Sodus Bay, Conesus Lake, Whitney Point Reservoir, and Onondaga Lake.

whether a lake is fishable. As previously shown in Table 2-2, these three ANC values correspond to levels at which distinct adverse effects on aquatic biota have been observed.

In the RUM framework, recreational fishing benefits accrue when lakes change from impaired (nonfishable) status to unimpaired (fishable) status. Therefore, applying the RUM framework in the RIA requires identifying the lakes that change from below the selected ANC threshold under the BAU scenario to above the selected threshold under the alternative regulatory scenarios.

It is also important to emphasize that the RUM benefit estimates depend on the spatial distribution of lakes that change status (i.e., *which* lakes improve) and not just on the number or percentage of lakes that change status. This is an inherent feature of the RUM framework, because it uses travel distances and travel costs to infer economic values.

Application of the Random Utility Model

To apply the RUM framework to estimate recreational fishing benefits, the primary input is the list of lakes that change status from impaired/nonfishable to unimpaired/fishable. The model's main benefit estimates are expressed as New York State residents' average willingness to pay (WTP) *in a given year* to improve recreational fishing services by reducing lake acidification levels. The data used to estimate the RUM were obtained from a 1989 repeat-contact telephone survey of New York residents; therefore, the estimates must be adjusted from 1989 dollars to more recent dollars (e.g., using the Consumer Price Index-All Urban Consumers [CPI-U]). These average annual WTP estimates can be generated for each year in the future, as long as predictions of lakes that change status in those years are available.

Estimation of Aggregate Benefits

To estimate aggregate benefits for New York residents, the per capita benefit estimates must be multiplied by the corresponding population of residents. To match the characteristics of the population surveyed in developing the RUM, the analysis requires estimates of the population of New York State that will be over 18 years old and reside outside of New York City for the future years of interest. These estimates can be acquired using the same Woods and Poole projections that are currently incorporated in EPA's BenMAP software (EPA, 2008a). These annual aggregate benefits can be estimated for 2020 and for subsequent years based on the modeling framework described above. The resulting stream of annual benefit estimates can then be expressed in present value terms for 2020 and also annualized using assumed discount rates of 3% and 7%.

2.2.1.3 Assumptions and Caveats

The following assumptions and caveats are particularly important for interpreting the results and the application of the ecological model for lake acidification:

- This analysis assumes that the level of impairment is binary as applied to a specific lake: that is, the ANC threshold indicates whether a lake is fishable. Consequently, it is not suited for estimating benefits that may accrue from changes that do not cross the specified thresholds.

- The available literature suggests that ANC levels between 20 and 100 cover the range where ecological effects are realized. Any point estimate within this range can be specified as the level at which the fishability of lakes is affected; therefore, it makes the most sense to conduct sensitivity analyses using alternative levels (e.g., 20, 50, and 100).

- This analysis assumed that the 35 modeled lakes are a representative subset of lakes in the Adirondacks (for the first tier of the analysis) and in New York State (for the second tier of the analysis).

- In the first tier, the analysis is not used to forecast acidification effects in Clinton, Saratoga, and Oneida counties, which are generally considered to be part of the Adirondack region because they are not represented by the subset of lakes subject to the ecological model. This restriction contributes to an underestimation of total benefits.

The following assumptions and caveats are particularly important for interpreting the application of the RUM model for estimating recreational fishing benefits to New York residents:

- The RUM only considers the behavior of New York State residents. It may be reasonable to assume that residents of neighboring jurisdictions may also take day trips to these lakes and respond in a rational manner comparable to New York State residents. This restriction contributes to an underestimation of benefits.

- This analysis assumed that the demand for fishing, in other words, an individual's propensity to fish, has remained constant from the time of the survey underlying the RUM to the present. That is, this analysis does not account for any potential change in interest in both recreational fishing and park use since the survey was conducted in 1989. In the case that general demand for recreational fishing has decreased, which is suggested by recent data from the FHWAR (DOI, 2007) for 1996 to 2006, this analysis may overstate benefits. This restriction contributes to an overestimation of benefits.

- This analysis did not take into account income adjustments through time. The RUM holds income to be constant, and a lack of detailed demand elasticity functions precludes the incorporation of an adjustment. Other EPA analyses have shown that increases in real income over time lead to increases in WTP for a wide range of health effects and some welfare effects, such as recreational visibility. This restriction contributes to an underestimation of benefits.

2.2.1.2 Total Ecological Benefits to New York Residents of Reducing Acidification of Adirondack Lakes

This section describes a second benefit estimation methodology that uses the Adirondack case study and MAGIC modeling results. This approach uses the results of a contingent valuation (CV) study conducted by researchers at Resources for the Future (RFF) (Banzhaf et al., 2006). The survey described and elicited values for specific improvements in acidification-related water quality and ecological conditions in Adirondack lakes; however, it specifically described the ecosystem services that would be enhanced by improving aquatic conditions. For this reason, and because the survey was administered to a random sample of New York households, the benefit estimates from the RFF study are interpreted as measures that incorporate values for *all* ecosystem services adversely affected by lake acidification (i.e., total ecological benefits).

Using the RFF study results, the fundamental benefit transfer model for valuing changes from the BAU baseline scenario to the regulatory alternative can be summarized as follows:

$$AggB_{lAdr} = WTP_{Adr} * N_{NY} * \Delta\%IL, \qquad (2.1)$$

where

$AggB_{Adr}$ = aggregate annual benefits (in constant dollars) to New York households in 2020 due to lake ecosystem improvements going from BAU to the regulatory alternative

WTP_{Adr} = average annual household WTP (in constant dollars) per unit of long-term change in the percentage of Adirondack lakes impaired by acidification

N_{NY} = projected total number of households in New York in 2020

$\Delta\%IL$ = long-term reduction in the percentage of Adirondack lakes impaired by acidification (going from BAU to the regulatory alternative)

First, results reported in Banzhaf et al. (2006) can be used to develop estimates of $WTP_{Adr,}$. The CV survey for the study was distributed to a random sample of nearly 6,000 New York residents in 2003 to 2004 through the Internet and mail. As part of the design and development of the survey instrument, experts were interviewed on the ecological damages, and a summary of the science was used as the foundation for describing the park’s existing condition and the hypothetical changes to be valued. The scientific review indicated that there was significant uncertainty regarding the future status of lakes in the Park in the absence of specific programs to improve lake acidification conditions. To bracket the range of uncertainty in the science as well as to test the sensitivity of respondents' WTP to the scale of ecological improvements, two versions of the survey instrument were developed and randomly administered to separate subsamples.

Table 2-4 summarizes key features of the two survey versions. In both survey versions, respondents were provided with information on the current (circa 2004) condition of the 3,000 lakes in the Park. Both versions describe half (1,500) of them as "lakes of concern" (i.e., unhealthy lakes where "fish and other aquatic life have been reduced or eliminated because of air pollution in the past"), and both versions propose policies that would improve the lakes over a period of 10 years (using lime to neutralize the excess acidity).

Table 2-4. Comparison of Resources for the Future Contingent Valuation Scenarios and EPA Zero-Out Scenario

	Percentage of Adirondack Lakes that Are "Unhealthy"			
		Future		
	Current (A)	No Program[a] (B)	With Program[b] (C)	Reduction (B) – (C)
RFF "Base" Scenario				
	Year = 2004	Year = 2014		
	50%	50%	30%	20%
RFF "Scope" Scenario				
	Year = 2004	Year = 2014		
	50%	55%	10%	45%

[a] Business-as-usual conditions.

[b] Lake liming program for the RFF survey scenarios and a zero-out policy for the EPA scenario.

The "base" version of the survey asserts that, in the absence of any direct policy intervention, the condition of the 1,500 unhealthy lakes and 1,500 healthy lakes is expected to remain unchanged over the next 10 years. However, if a liming program is undertaken, it would improve 20% (600) of the lakes in the Park relative to their expected 2014 condition without the program.

In contrast, the "scope" version describes a gradually worsening status quo without the liming program, in which 5% (150) of the healthy lakes are expected to gradually become unhealthy. In other words, without the program, 55% (1,650) of the lakes would be unhealthy in

2014. With the liming program, however, only 10% of the lakes would be unhealthy in 2014, so the program improves 45% (1,350) of the lakes relative to their expected 2014 condition without the program.

Although scientific evidence indicates that it is uncertain whether a liming policy would significantly improve the condition of birds and forests, pretesting of the survey indicated that respondents may nonetheless have tended to assume that these other benefits would occur. Therefore, to make the scenarios more acceptable to respondents, other nonlake effects were added to the two survey versions. In the base case, the red spruce (covering 3% of the forests' area) and two aquatic bird species (common loon and hooded merganser) are said to be affected. In this version, the health of birds and forests is described as unchanged in the absence of an intervention, and minor improvements are said to result from the program. In the scope version, a broader range of damages is associated with acid rain—two additional species of trees (sugar maple and white ash, all together covering 10% of forest area) and two additional birds (wood thrush and tree swallow) are said to be affected. The scope version describes a gradually worsening status quo along with large improvements due to the program.

Respondents were presented with one of these (base or scope) policy scenarios and then asked how they would vote in a referendum on the program, if it were financed by an increase in state taxes for 10 years. To estimate the distribution of WTP, the annual tax amounts were randomly varied across respondents.

Based on a detailed analysis of the survey data, Banzhaf et al. (2006) defined a range of best WTP estimates, which were converted from 10-year annual payments to permanent annual payments using discount rates of 3% and 5%. For the base version, the best estimates ranged from $48 to $107 per year per household (in 2004 dollars), and for the scope version they ranged from $54 to $154. If a 7% annual discount rate is used to convert the 10-year annual payments to permanent annual payments, then the upper end of these ranges increases to $149 and $216, respectively.

To specify values for WTP_{Adr}, these estimates can be converted to more recent 2007 dollars using the CPI and divided by the corresponding change in the percentage of lakes that are unhealthy (20% for the base version and 45% for the scope version). For the base version, the WTP_{Adr} estimates range from $2.63 to $8.81 per percentage decrease in unhealthy lakes, and for the scope version they range from $1.32 to $5.26. Selecting the best range of values to use for WTP_{Adr} depends mainly on which scenario—the base or the scope version—corresponds most closely with the predicted changes in the percentage of impaired Adirondack lakes under the BAU and regulatory scenarios ($\Delta\%IL$).

To estimate N_{NY}, the Woods and Poole population projection from BenMAP can again be used.

To estimate $\Delta\%IL$, it is again necessary to extrapolate MAGIC modeling results to the larger universe of Adirondack lakes. However, in contrast to the RUM framework described in Section 2.2.1.1, the benefit transfer model summarized in Equation (2.1) is not sensitive to the spatial distribution of lake status changes. Rather, it depends strictly on the percentage of lakes that are impaired under the different scenarios. Consequently, a lake-specific extrapolation approach, such as the previously described random effects model application, is not necessarily needed.

The key issue in extrapolating the MAGIC model results to define $\Delta\%IL$ is whether and to what extent the 44 MAGIC modeled lakes represent (or can be adjusted to represent) ANC conditions across all Adirondack lakes. This issue is partially addressed in the REA, which compares modeled results to monitored ANC values from a probability sample of 94 lakes. The comparison, which is shown in Table 2-5, suggests that the distributions are roughly similar, but the modeled lakes tend to be in the lower concern (higher ANC) categories.

Table 2-5. Percentage of Lakes in the Five Aquatic Status Categories Based on Their Surface Water ANC Concentrations for 44 Lakes Modeled Using MAGIC and 94 Lakes in the TIME/LTM Monitoring Network. Results Are for the Adirondack Case Study Area for the Year 2006.

Concern	ANC (μeq/L)	Modeled Current Condition (% of Lakes)	Measured Current Condition (% of Lakes)
Low	>100	20	6
Moderate	50–100	36	16
Elevated	20–50	23	32
Severe	0–20	9	29
Acute	<0	11	17

Source: EPA (2009)

2.2.1.3 Limitations and Uncertainties

First, uncertainties are associated with extrapolating results from the 44 MAGIC-modeled lakes to all (roughly 3,000) Adirondack lakes. The 44 modeled lakes are drawn from a larger, randomly drawn sample of lakes; however, the representativeness of these 44 lakes for the Adirondacks as a whole is uncertain.

Second, there is also some uncertainty related to the exact types of ecosystem services that are included in these RFF study values, particularly regarding provisioning and regulating services (e.g., fish consumption and forest production services or biological control services), which survey respondents may have been less likely to consider when formulating responses to the CV questions. Importantly though, the values estimated by the RFF study are likely to include recreational fishing services, which means *they cannot be added to the RUM results*, and other cultural services, in particular, recreational and nonuse services.

Third, the inclusion of other ecosystem changes (e.g., trees, birds) in the RFF CV survey scenarios implies that respondents' stated values will overstate WTP for just changes in lake acidification. That is, they may also include some benefits from reductions in terrestrial acidification. This feature, therefore, contributes to potential overestimation of benefits.

Fourth, the lack of direct correspondence between the RFF CV scenarios and the regulatory scenarios requires assumptions for making the benefit transfer. In addition to differences in the timing of lake improvement changes and, potentially, differences in the baseline percentage of impaired lakes, there are also likely to be differences in the percent of lakes improved. Rescaling the WTP estimates for different percentage changes in unhealthy lakes requires the assumption that there is a constant WTP *per percentage* decline in unhealthy lakes. Unfortunately, neither these survey results nor results from other studies provide strong guidance regarding the appropriateness of this assumption relative to alternative assumptions.

Finally, the results from this method only apply to Adirondack lakes and to New York residents. The Adirondack region is more sensitive to acidity in contrast to many other areas of New York State, which have calcium-rich limestone deposits that neutralize the acid. The bedrock soil and shallow soil deposits have a lower buffering capacity. These geological factors together with high acidic precipitation levels contribute to the vulnerability of this region to acidification. The uniqueness of the Park makes simple extrapolations of ecological conditions and human values to other lakes very uncertain. Similarly, residents of other states are likely to value improved ecosystem services from Adirondack lakes, but the magnitude of these values is difficult to assess. The omission of these values contributes to an underestimation of benefits.

2.2.2 *Benefits of Reduced Acidification in Shenandoah Streams*

The Shenandoah case study uses a weighting scheme based on the bedrock class of the 60 modeled streams to extrapolate conditions and changes to 310 brook trout streams in the Shenandoah. Using geographical information system (GIS) data on bedrock type and on the location, number, and characteristics of streams, it may be possible to further geographically extrapolate these results to other streams in the larger Blue Ridge region, although doing so would involve greater amounts of uncertainty.

In contrast to the Adirondack case study, no valuation studies exist that specifically address changes in acidification-related water quality in the Shenandoah region.[4] Nevertheless, an alternative benefit transfer approach can be applied using results from another EPA-funded stated preference (SP) study (Viscusi, Huber, and Bell [VHB], 2008a).[5] Using a nationwide sample, this study elicited values for regional water quality changes that were not specific to a particular area of the country. Consequently, the study results offer flexibility and can applied in many different areas, including the Shenandoah.

The VHB survey study was conducted from 2002 to 2004, using a nationwide sample of 4,257 respondents from the Knowledge Networks WebTV household panel. The main SP task in this survey presented individuals with a hypothetical scenario in which they planned to move and had a choice between two regions (100-mile radius each). The regions only differ with respect to the increase in annual cost of living and the percentage of lakes and rivers that are “good” water quality. Respondents were told that waterbodies are classified as good if (1) the fish is safe to eat; (2) the water is safe to swim in; *and* (3) the water supports a healthy environment for fish, plants, and other aquatic life. By varying the cost-of-living differences and water quality differences and observing how respondents’ choices vary in response to these changes, the study estimated an average (median) annual household WTP of $32 ($13) per 1% increase in good waters. Another SP task asked individuals to make similar types of trade-off choices where, instead, the type of waterbody (river or lake) and the type of quality indicator (safe for fish consumption, swimming, or aquatic life) were varied. The results indicated that individuals are willing to pay slightly more for higher lake than higher river quality (53.1% of WTP compared to 46.9%) and slightly less for safe swimming (30% of WTP) compared to safe fish consumption

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[4]A valuation study addressing surface water acidification in the broader Appalachian region is currently being developed and implemented (http://cfpub1.epa.gov/ncer_abstracts/index.cfm/fuseaction/display.abstractDetail/abstract/7726); however, no results from this study are available yet.

[5]Other publications based on this study include Viscusi, Huber, and Bell (2008b) and Huber, Viscusi, and Bell (2006, 2008). A pilot version of this study using a convenience sample is discussed in Magat et al. (2000).

(35.2%) and safe for aquatic life (34%). Past participation in freshwater recreation, income, age, and education were all found to have significantly positive effects on WTP, whereas household size had a negative effect.

The VHB study also provides and demonstrates a regression-based valuation (i.e., benefit transfer) function that can be used to estimate WTP for selected water quality improvements defined as the change in percentage of "good" quality freshwater. Before applying this function to estimate the benefits of reductions in freshwater acidification for a specific control scenario, it is important to define the spatial scope of the analysis.

The first step is to define the geographic region over which water quality changes will be estimated and valued. The modeling of ANC levels under alternative deposition scenarios for the Shenandoah region was applied to 60 individual trout streams. Although it is difficult to determine the exact extent to which these results can be spatially extrapolated to other waterbodies and areas, the case study does conclude that "[t]he 60 trout streams modeled are characteristic of first- and second-order streams on nonlimestone bedrock in the Blue Ridge Mountains of Virginia" (EPA, 2009). In addition, it states that "[b]ecause of the strong relationship between bedrock geology and ANC in this region, it is possible to consider the results in the context of similar trout streams in the Southern Appalachians that have the same bedrock geology and size."

Using the Virginia Blue Ridge Mountains (VABRM) as the relevant study area for benefits analysis, Figure 2-3 shows the boundaries of this region, which correspond rather closely to the Shenandoah case study area. This region was defined using EPA's Level III ecoregions to identify the Blue Ridge Mountain areas of Virginia. The study area was extended into West Virginia to capture the entire ecoregion. The northern border of the study area follows the boundary of Virginia from the northeast corner of the study area and then joins the boundary of West Virginia in the northwest corner of the study area. The southern border follows the Virginia state boundary alone.[6] The enhanced National Hydrography Data set (NHDPlus) was used to define the hydrography of the area, which is also shown in the map.

The second step is to define an approach for extrapolating estimates from the MAGIC-modeled streams to all rivers and streams in the study area. Based on the Shenandoah case study

[6]Using the Virginia state borders as a boundary is somewhat arbitrary; however, including the entire Blue Ridge and Valley and Ridge region might extend the analysis into areas where deposition levels are not adequately represented by the existing MAGIC results for the Shenandoah region.

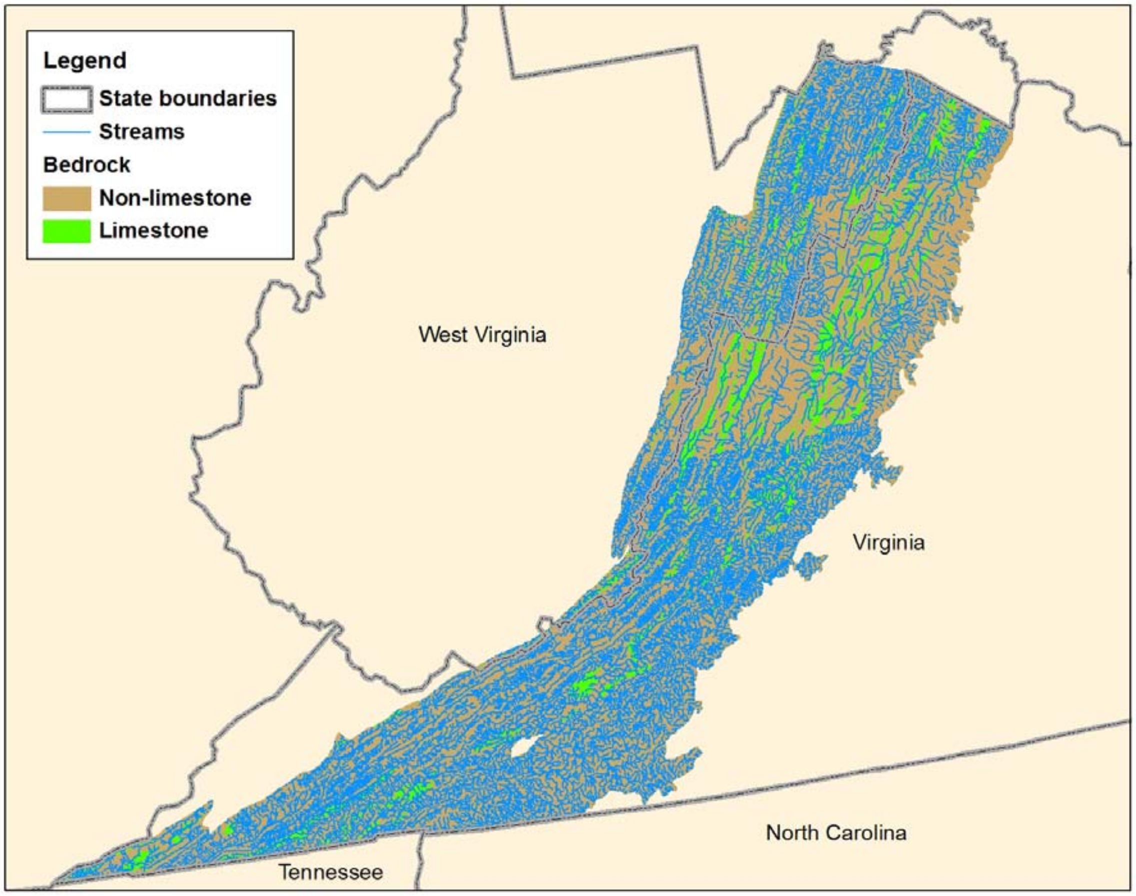

Figure 2-3. Virginia Blue Ridge Mountain Study Area: Stream Network and Bedrock Type

results reported in the REA, the first column in Table 2-5 reports the percentage of the modeled streams that are "impaired" according to the three ANC thresholds. All of the modeled streams are in areas without limestone bedrock. Limestone bedrock is not conducive to acidification; therefore, ANC values in limestone areas are assumed to be above 100 μeq/L. To extrapolate these results to other rivers and streams, we assumed that the percentage of impaired streams in nonlimestone bedrock is the same for all first, second, and higher order streams in the VABRM area. As shown in Table 2-6, almost 80% of streams in this area are in the first and second order

Table 2-6. Extrapolation of MAGIC-Modeled ANC Levels to Other Rivers and Streams in the VA Blue Ridge Mountain Study Area

	MAGIC-Modeled Streams	First and Second Order Streams	Other Higher Order Rivers/Streams	All Rivers and Streams
Number of rivers/streams	60 streams	26,874 miles	7,005 miles	33,880 miles
Percentage on nonlimestone bedrock	100%	90%[a]	88%[a]	89%[a]
	Modeled Estimates	**Extrapolated Estimates[b]**		
Percentage below ANC=20 μeq/L	25%	17%	15%	16%
Percentage below ANC=50 μeq/L	53%	48%	47%	47%
Percentage below ANC=100 μeq/L	73%	70%	69%	70%

[a] Source: NHDPlus

[b] Adjusted to account for percentage of streams on limestone bedrock, which are assumed to be above ANC=100.

(i.e., smallest) size categories and roughly 89% of all streams are located on nonlimestone bedrock.[7]

The third step is to match the changes in stream conditions with proximate populations. This linkage is needed for at least two reasons. First, the average characteristics of these populations (e.g., income, age) can be used in the valuation function to adjust predictions of the *average* (per capita) WTP. Second, the size of the affected population is needed to calculate *aggregate* benefits. To be completely consistent with the VHB valuation, separate areas (100-mile radius circles) and corresponding percentage changes in "good" water should be defined and calculated for each geographically distinct population (e.g., Census block) within 100 miles of the affected waters. However, such an approach is likely to be computationally overly burdensome.

A more limited but also more feasible approach is to select the population living within a specified distance of the VABRM study area and use the size and average characteristics of this

[7]The classifications of first, second, and other ordered stream was based on Stahler stream order data, which are available for use in conjunction with the NHDPlus. The bedrock classifications were produced from a national geologic layer from http://tin.er.usgs.gov/geology/state/. It should be noted that there were approximately 560 km of stream segments within the study area without designated stream orders. This small portion (2%) of stream segments typically represented diversions or uninitialized stream segments and, therefore, can be disregarded in the analysis. An even smaller percentage (0.3% or 94 km) of stream segments were disregarded during the bedrock overlay from the small area that had a bedrock listing of water.

population to estimate average WTP. For example, the population could be limited to individuals living within the VABRM study boundary (as shown in Figure 2-3). This approach would most likely underestimate aggregate benefits and, therefore, provide a lower-bound estimate, because it would exclude individuals living outside the study area (e.g., up to 100 miles) who would benefit from the improvement. However, it would also be likely to overstate WTP for individuals living closer to the edges of the study area.[8] A second alternative would be to select the population living within 100 miles of the affected waters. This approach would most likely overstate aggregate benefits and, thus, provide an upper bound.

The final step is to define and apply the benefit transfer model based on the VHB study results. The fundamental model for valuing changes from the BAU baseline scenario to the regulatory alternative can be summarized as follows:

$$AggB_{BR} = WTP_{VHB} * N_{BR} * \Delta\%IR * (0.469), \tag{2.2}$$

where

$AggB_{BR}$ = aggregate annual benefits (in constant dollars) to households in the VABRM region in 2020 due to river ecosystem improvements going from BAU to the regulatory alternative

WTP_{VHB} = VHB estimate of the average annual household WTP (in constant dollars) per unit change in the percentage of impaired (i.e., without "good" water quality) lakes and rivers in the region

N_{BR} = projected total number of households in and proximate to the VABRM region in 2020

$\Delta\%IR$ = reduction in the percentage of rivers and streams impaired by acidification (going from BAU to the regulatory alternative). The coefficient 0.469 represents the share of WTP_{VHB} allocated to improvements in rivers (rather than lakes)

To estimate $\Delta\%IR$, it is assumed that MAGIC-modeled estimates for 2020 will be generated for the BAU and regulatory alternative scenarios. These estimates can then be extrapolated to all rivers and streams in the VABRM area using the same approach summarized in Table 2-5.

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[8]The radius for these individuals would include more waters outside the study area, where improvements are presumably lower.

The average WTP_{VHB} estimate for the national sample used in the VHB study is reported as \$31.70 (in 2004 dollars); however, this estimate can also be tailored to the demographic and geographic characteristics in the Shenandoah region. Using the regression equation reported in Huber, Viscusi, and Bell (2006), this average WTP can be estimated as follows:

$$WTP_{VHB-BR} = \exp\begin{bmatrix} 0.667 + (0.124 * \ln Inc) + (0.0392 * Educ) + (0.0062 * Age) \\ + (0.5275 * Envorg) + (0.1930 * Visit) - (0.127 * Black) \\ + (0.0165 * Other) + (0.1077 * Hisp) - (0.0478 * Fem) \\ - (0.0294 * HHsize) - (0.04 * South) + (0.0004 * LakeWQ) \\ + (0.0044 * Lakedens) + 0.4332 \end{bmatrix} \tag{2.3}$$

where

lnInc = natural log of average household annual income (in 2004 dollars)

Educ = average number of years of education

Age = average age (in years)

Envorg = percentage of population member of environmental organization

Visit = percentage of population who visited lake or river in last 12 months

Black = percentage of population who are black

Hisp = percentage of population who are Hispanic

Other = percentage of population who neither black, white, or Hispanic

Fem = percentage of population who are female

HHsize = average household size (number of people)

South = Indicator variable =1 if region is in the South

LakeWQ = percentage of good quality lakes in state

Lakedens = lake acres per square mile in state

Most of these explanatory variables can be estimated using Census data and projections for the selected counties in the defined Shenandoah region. For other variables such as *EnvOrg* and *Visit*, Huber, Viscusi, and Bell (2006) recommend using averages from their sample (*Envorg* = 54% and *Visit* = 67.4%) as a default. According to most recent data from EPA's Assessment

Total Maximum Daily Load (TMDL) Tracking and Implementation System [ATTAINS] (http://www.epa.gov/waters/ir/) for VA, *LakeWQ*= 16.2% and *Lakedens* = 3.51 acres/sq mi.

These annual aggregate benefits can be estimated for 2020 and for subsequent years based on the available MAGIC results and the modeling framework described above. The resulting stream of annual benefit estimates can then be expressed in present value terms for 2020 and also annualized using assumed discount rates of 3% and 7%.

2.2.1.1 Limitations and Uncertainties

One of the main limitations of this approach is the uncertainty associated with spatially extrapolating water quality modeling results from 60 modeled trout streams in the Shenandoah to a larger geographic area and other types and orders of streams. As discussed above, the REA concludes that the modeled streams are characteristic of most streams in the Blue Ridge Mountain region of Virginia; nevertheless, the direct extrapolation of these results to all rivers and streams is an important source of uncertainty.

A second source of uncertainty arises from the definition of "good" water quality. As stated above, survey respondents were told that waterbodies are classified as good if (1) the fish is safe to eat; (2) the water is safe to swim in; *and* (3) the water supports a healthy environment for fish, plants, and other aquatic life. To apply the VHB study, we use alternative ANC thresholds to define waters with good quality. In addition to the uncertainty about which of these thresholds—20, 50 or 100 μeq/L—is most appropriate, acidification levels only affect one of the three water quality dimensions (healthy aquatic life) and other stressors may also affect this dimension. Consequently, to the extent that other types and sources of impairment are limiting factors, using only modeled ANC levels to characterize the change in good quality waters ($\Delta\%IR$) may result in overestimates of improved waters and of the benefits of reduced acidification. Data measuring other types water quality impairments in the VABRM (e.g., from EPA's Wadeable Streams Assessment (EPA, 2006) can help characterize this potential overestimation.

A third source of uncertainty is the specification of benefiting population—$N_{BR.}$ As described above, alternative assumptions regarding the spatial boundaries for this population can help bound the aggregate benefit estimates. According to Census data, 516,000 households lived within the VABRM boundary in 2000, compared to over 8.6 million within 100 miles of the boundary. Similar estimates can be derived for future years using the Woods and Poole projections that are also included in BenMAP.

2.3 Summary of Modeling Framework

Figure 2-4 summarizes the main proposed modeling steps for assessing the benefits of reductions in aquatic acidification due to N+S deposition. As in the discussion above, it uses the two case study applications—Adirondacks lakes and Shenandoah streams—to illustrate these modeling steps. To the extent that MAGIC modeling results become available for other areas, the methods described in particular for the VABRM benefits analysis can be adapted to estimate benefits for those areas as well.

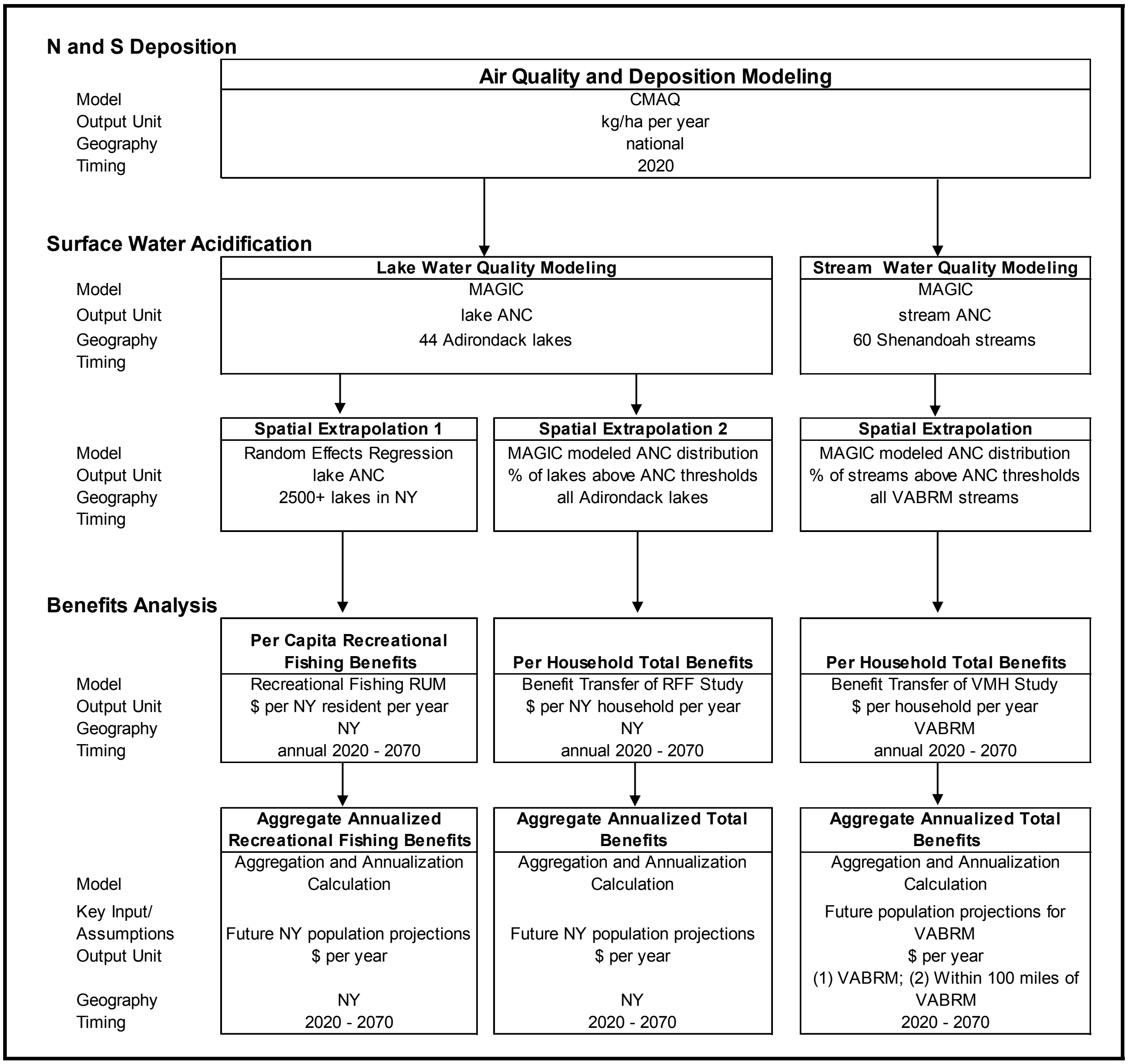

Figure 2-4. Key Modeling Steps for Assessing the Benefits of Reduced Aquatic Acidification

2.4 References

Banzhaf, S., D. Burtraw, D. Evans, and A. Krupnick. 2006. "Valuation of Natural Resource Improvements in the Adirondacks." *Land Economics* 82:445-464.

Chen, L., and C.T. Driscoll. 2004. "Modeling the Response of Soil and Surface Waters in the Adirondack and Catskill Regions of New York to Changes in Atmospheric Deposition and Historical Land Disturbance." *Atmospheric Environment* 38:4099-4109.

Driscoll, C. T, G. B. Lawrence, A. J. Bulger, T. J. Butler, C. S. Cronan, C. Eagar, K. F. Lambert, G. E. Likens, J. L. Stoddard, and K. C. Weathers. 2001. "Acidic Deposition in the Northeastern United States: Sources and Inputs, Ecosystem Effects, and Management Strategies." *BioScience* 51(3):180–198.

Driscoll, C.T. et al. 2003. "Effects of Acidic Deposition on Forest and Aquatic Ecosystems in New York State." *Environmental Pollution* 123:327-336.

Huber, J., W.K. Viscusi, and J. Bell. 2006. *Economics of Environmental Improvement*. Report prepared under EPA Cooperative Agreement CR823604 and Grant R827423.

Huber, J., W.K. Viscusi, and J. Bell. 2008. "Reference Dependence in Iterative Choices, Organizational Behavior and Human Decision Processes." Available at http://www.sciencedirect.com/science/article/B6WP2-4RPKYMH-1/1/31461eb21bfe85962fe5672b831f9127.

Hutchison, R., and C.E. Kraft. 1994. "Hmong Fishing Activity and Fish Consumption." *Journal of Great Lakes Research* 20(2):471-487.

Industrial Economics, Inc. June 2008. "The Economic Impact of the Clean Air Interstate Rule on Recreational Fishing in the Adirondack Region of New York State." Prepared for the Clean Air Markets Division, Office of Air and Radiation, U.S. EPA.

Kaval, P., and J. Loomis. 2003. *Updated Outdoor Recreation Use Values with Emphasis on National Park Recreation*. Final Report under Cooperative Agreement CA 1200-99-009, Project number IMDE-02-0070.

Magat, W.A., J. Huber, W.K. Viscusi, and J. Bell. 2000. "An Iterative Choice Approach to Valuing Clean Lakes, Rivers, and Streams." *Journal of Risk and Uncertainty* 21(1):7-43.

Millennium Ecosystem Assessment (MEA). 2005. Ecosystems and Human Well-being: Wetlands and Water. Synthesis. A Report of the Millennium Ecosystem Assessment. Washington, DC: World Resources Institute.

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

Peterson, D.E., M.S. Kanarek, M.A. Kuykendall, J.M. Diedrich, H.A. Anderson, P.L. Remington, and T.B. Sheffy. 1994. "Fish Consumption Patterns and Blood Mercury Levels in Wisconsin Chippewa Indians." *Archives of Environmental*

U.S. Environmental Protection Agency (EPA). December 2006. *Wadeable Streams Assessment: A Collaborative Survey of the Nation's Streams*. EPA 841-B-06-002. Washington, DC: EPA. Available at http://www.epa.gov/owow/streamsurvey/.

U.S. Environmental Protection Agency (EPA). 2008a. *BenMAP: Environmental Benefits Mapping and Analysis Program: User's Manual*. RTP, NC: EPA. Available at http://www.epa.gov/air/benmap/models/BenMapManualSept08.pdf.

U.S. Environmental Protection Agency (EPA). 2008b. *Integrated Science Assessment (ISA) for Oxides of Nitrogen and Sulfur–Ecological Criteria (Final Report).* EPA/600/R-08/082F. U.S. Environmental Protection Agency, National Center for Environmental Assessment–RTP Division, Office of Research and Development, Research Triangle Park, NC. Available at http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=201485.

U.S. Environmental Protection Agency (EPA). 2009. *Risk and Exposure Assessment for Review of the Secondary National Ambient Air Quality Standards for Oxides of Nitrogen and Oxides of Sulfur.* Research Triangle Park, NC: U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards. EPA-452/R-09-008a.

Second Section 812 Prospective Project Team. February 2007. "Assessing the Effects of the Clean Air Act Amendments of 1990 on Ecological Resources: Updated Literature Review and Terrestrial Case Study Approach." White Paper. Available at http://www.epa.gov/air/sect812/mar07/eco_assessment.pdf.

Viscusi, W.K., J. Huber, and J. Bell. 2008a. "The Economic Value of Water Quality." *Environmental and Resource Economics* 41:169-187.

Viscusi, W.K., J. Huber, and J. Bell. 2008b. "Estimating Discount Rates for Environmental Quality from Utility-Based Choice Experiments." *Journal of Risk and Uncertainty*. DOI 10.1007/s11166-008-9045-x.

SECTION 3
TERRESTRIAL ACIDIFICATION

3.1 Overview of Adversely Affected Ecosystems and Services

Terrestrial acidification is the result of natural processes and anthropogenic sources of acidic deposition. Figure 3-1 depicts the geographical location of the acidic deposition areas in the United States. As can be seen from the figure, high N+S deposition areas are largely concentrated in the northeastern region.

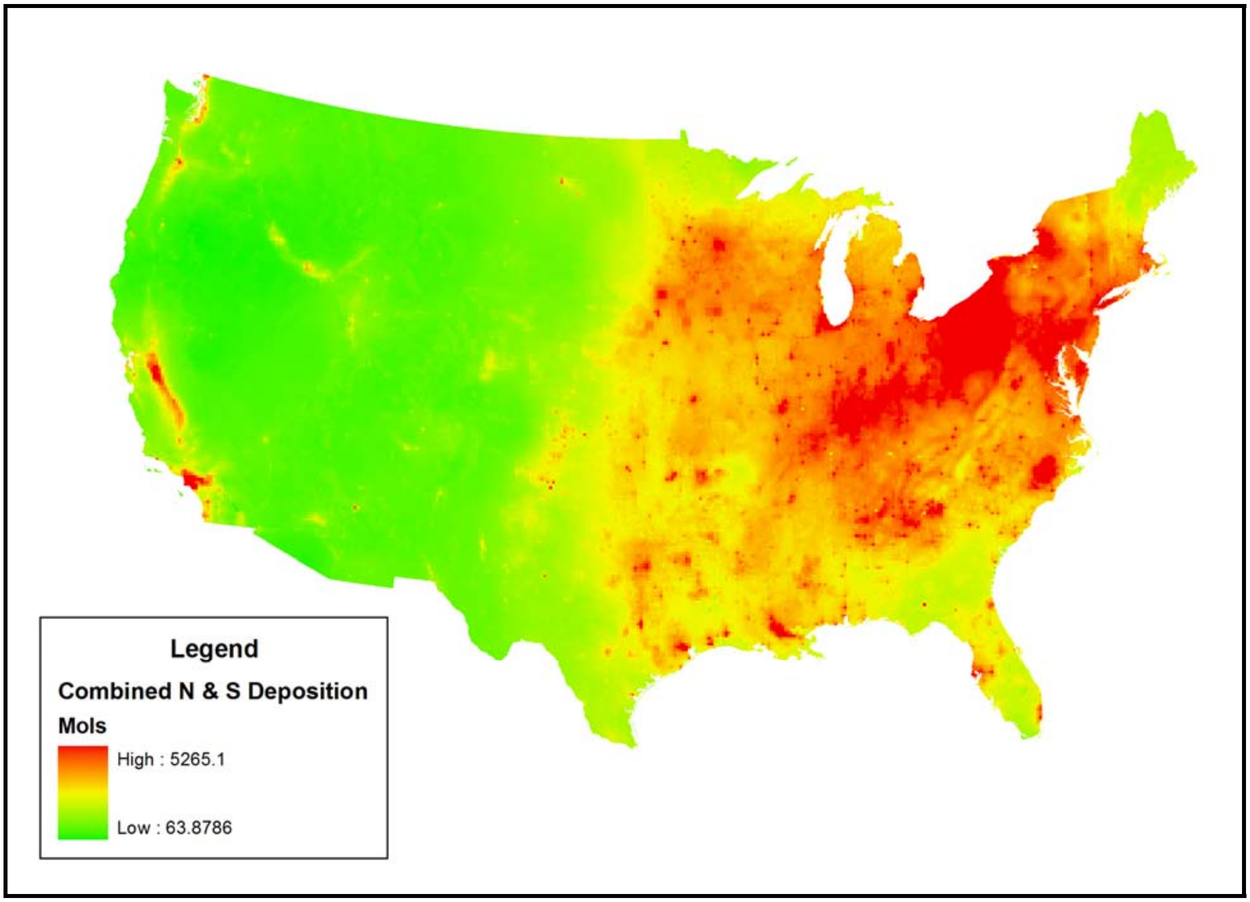

Figure 3-1. Combined N+S Deposition (from 2002 CMAQ Dry Deposition and NADP Wet Deposition Estimates)

In addition to causing adverse effects in aquatic ecosystems, as described in the previous section, elevated levels of atmospheric deposition of N+S can cause a variety of damages to terrestrial ecosystems. Figure 3-2 provides a conceptual framework tracing out the main

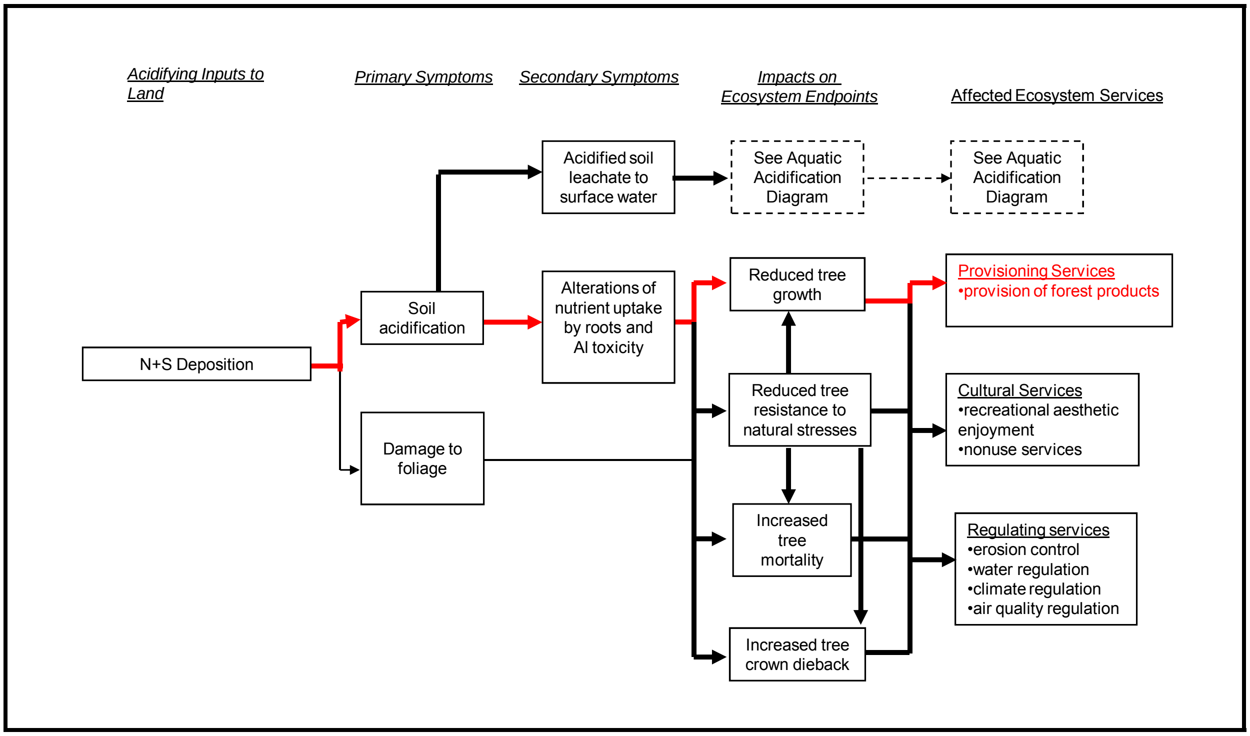

Figure 3-2. Conceptual Diagram of Ecosystem Service Impairments Associated with Terrestrial Acidification

pathways through which acidic deposition adversely affects these ecosystems and impairs the services they provide to humans.

When inputs of N+S from atmospheric deposition exceed levels that can be naturally buffered by soils, they alter the chemical composition of soils by accelerating rates of base cation (e.g., calcium and magnesium) leaching. This acidification process in soils depletes available plant nutrients and contributes to the mobilization and leaching aluminum, which can be toxic to tree roots. At the same time, deposition of N+S to the surface of trees can cause direct damage to tree foliage. Consequently, damages to tree health are among the most visible and significant effects of acid deposition. These damages are exhibited through slower growth of tree biomass and decreased resistance to other natural and manmade stresses. In more extreme cases, acidification may lead to increased tree crown dieback and mortality.

3.1.1 Effects on Provisioning Services

Forests in the United States provide several important and valuable provisioning services, which are reflected in measures of production and sales of tree products. Tree health and growth may be compromised both directly and indirectly by acidifying N+S deposition due to induced soil nutrient deficiencies and imbalances caused by the leaching of base cations from the soil. Tree growth may be reduced and/or trees may have an increased susceptibility to drought and pest damage, aluminum (Al) toxicity in roots, reduced tolerance to cold, and a greater propensity to frost injury (Driscoll et al., 2001; DeHayes et al., 1999; Fenn et al., 2006; McNulty et al., 2005; Ouimet et al., 2008). In addition, tree mortality may increase due to the stresses induced by N+S deposition (DeHayes et al., 1999; Drohan and Sharpe, 1997; Perkins et al., 2000; Schaberg et al., 2000). This reduced growth and increased tree mortality may consequently result in reduced sales of tree products.

Evidence of adverse effects due to terrestrial acidification is particularly strong for two common tree species in the northeastern United States where levels of N+S deposition have historically been relatively high—sugar maples and red spruce. Therefore, these two species provide useful examples of the types and magnitude of provisioning services adversely affected by terrestrial acidification.

Sugar maple is a particularly important commercial hardwood tree species in the United States. As shown in Figure 3-3, its range covers most of the United States east of the Mississippi River and north of Alabama and Georgia, which is also the area with highest levels of N+S deposition in the country. In addition to being the source of maple syrup, wood from sugar maple is widely used in construction, furniture, and flooring (Luzadis and Gossett, 1996). According to data from the U.S. Forest Service's National Forest Inventory and Analysis (FIA) databases (http://199.128.173.26/fido/mastf/index.html), in 2006, the total removal of sugar maple saw timber from timberland in the United States was almost 900 million board feet. Assuming a range of sugar maple prices from \$100 to \$1,000[1] per 1,000 board feet, the total value of these removals in 2006 ranged between \$90 and \$900 million. Annual revenues from maple syrup production in the United States averaged roughly \$160 million for 2005 to 2007.

Red spruce is a common commercial softwood species, which is now mainly found in northern New England, New York, and in a few high-elevation areas of the Appalachian Mountain range. Wood from red spruce is used in a variety of products including lumber, pulpwood, poles, plywood, and musical instruments. According to FIA data, in 2006, the total

[1]These prices are available at http://www.srs.fs.usda.gov/econ/data/prices/index_t.htm. Prices outside this range are also reported but these seem to be more uncommon. The wide range reflects seasonal and regional variation as well as differences in quality and market conditions.

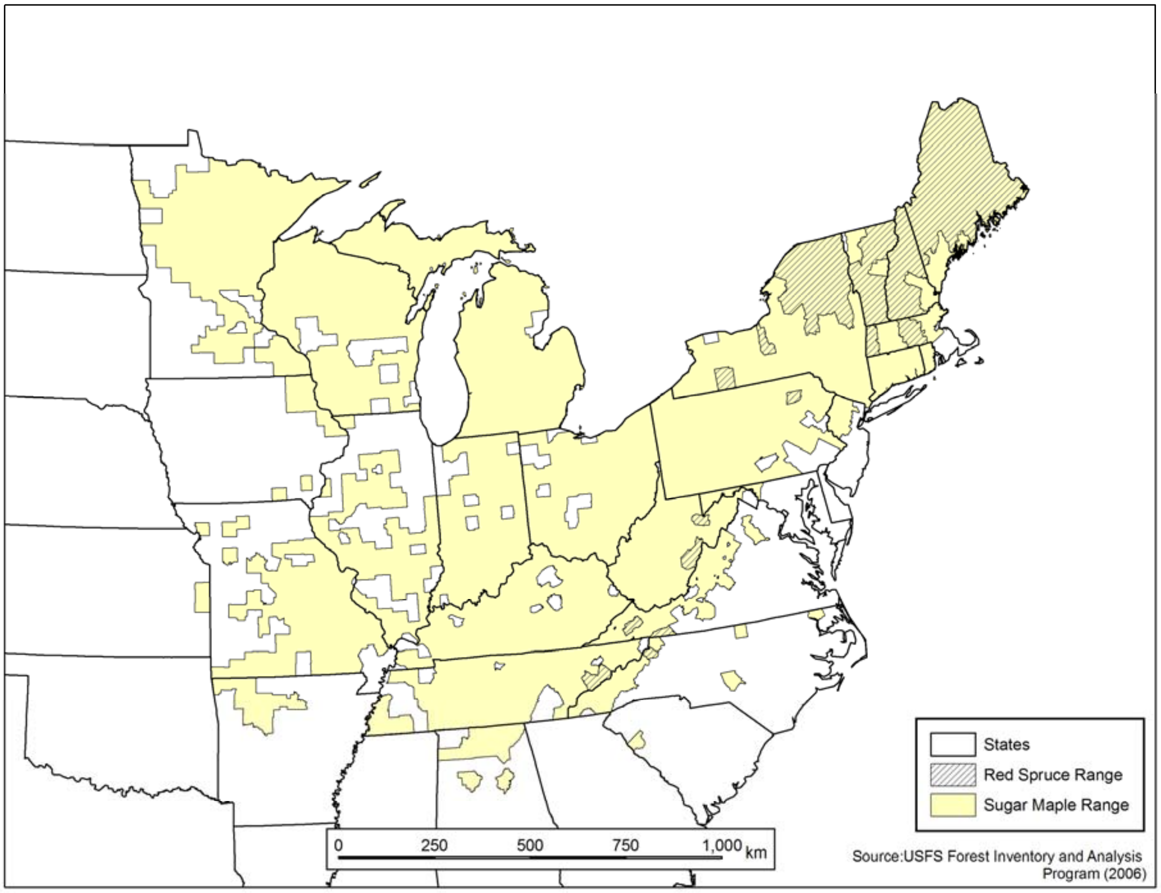

Figure 3-3. Areal Coverages of Red Spruce and Sugar Maple Tree Species within the Continental United States (USFS, 2006)

removal of red spruce saw timber from timberland in the United States was 328 million board feet. Assuming a range of red spruce prices from $25 to $300 per 1,000 board feet, the total value of these removals in 2006 ranged between $8 and $99 million.

3.1.2 Effects on Cultural Services

Forests in the northeastern United States are also an important source of cultural ecosystem services—in particular recreational and aesthetic services. Forest lands support a wide variety of outdoor recreational activities, including fishing, hiking, camping, off-road driving, hunting, and wildlife viewing. Regional statistics on recreational activities that are specifically forest based are not available; however, more general data on outdoor recreation provide some insights into the overall level of recreational services provided by forests. For example, most recent data from the National Survey on Recreation and the Environment (NSRE) indicate that, from 2004 to 2007, 31% of the U.S. adult (16 and older) population visited a wilderness or primitive area during the previous year, and 32% engaged in day hiking (Cordell et al., n.d.). From 1999 to 2004, 16% of adults in the northeastern United States[2] participated in off-road vehicle recreation, for an average of 27 days per year (Cordell et al., 2005). Using the meta-analysis results reported by Kaval and Loomis (2003), which found that the average consumer surplus value per day of off-road driving in the United States was $25.25 (in 2007 dollars), the implied total annual value of off-road driving recreation in the northeastern United States was more than $9.25 billion.

State-level data on other outdoor recreational activities associated with forests are also available from the 2006 FHWAR (U.S. Department of the Interior [DOI], 2007). As summarized in Table 3-1, 5.5% of adults in the northeastern United States, which covers a majority of the high deposition areas, participated in hunting, and the total number of hunting days occurring in those states was 83.8 million. Data from the survey also indicated that 10% of adults in northeastern states participated in wildlife viewing away from home. The total number of away-from-home wildlife viewing days occurring in those states was 122.2 million in 2006. For these recreational activities in the northeastern United States, Kaval and Loomis (2003) estimated average consumer surplus values per day of $52.36 for hunting and $34.46 for wildlife viewing (in 2007 dollars). The implied total annual value of hunting and wildlife viewing in the northeastern United States was, therefore, $4.38 billion and $4.21 billion, respectively, in 2006.

As previously mentioned, it is difficult to estimate the portion of these recreational services that are specifically attributable to forests and to the health of specific tree species. However, one recreational activity that is directly dependent on forest conditions is fall color viewing. Sugar maple trees, in particular, are known for their bright colors and are, therefore, an essential aesthetic component of most fall color landscapes. Declines in sugar maple stocks due to terrestrial acidification are, thus, expected to have detrimental effects on these landscapes. Statistics on fall color viewing are much less available than for the other recreational and tourism activities; however, a few studies have documented the extent and significance of this activity. For example, based on a 1996 to 1998 telephone survey of residents in the Great Lakes area, Spencer and Holecek (2007) found that roughly 30% of residents reported at least one trip in the previous year involving fall color viewing. In a separate study conducted in Vermont, Brown

[2]This area includes Connecticut, Delaware, District of Columbia, Illinois, Indiana, Maine, Maryland, Massachusetts, Michigan, New Hampshire, New Jersey, New York, Ohio, Pennsylvania, Rhode Island, Vermont, West Virginia, and Wisconsin.

Table 3-1. Participation in Hunting and Wildlife Viewing in the Northeastern (i.e., High Deposition) Region of the United States in 2006

State	Participation Rates by State Residents[a]		Activity Days by Residents and Nonresidents (in thousands)	
	Hunting	Wildlife Viewing[b]	Hunting	Wildlife Viewing[b]
Connecticut	1.2%	11.0%	509	4,184
Delaware	3.1%	7.0%	654	855
Illinois	2.8%	8.0%	4,688	5,686
Indiana	5.3%	13.0%	4,808	24,013
Maine	13.6%	20.0%	2,283	4,778
Maryland	3.5%	7.0%	2,262	4,782
Massachusetts	1.3%	11.0%	1,149	8,461
Michigan	9.2%	11.0%	11,905	10,043
New Hampshire	5.0%	12.0%	1,057	3,165
New Jersey	1.3%	8.0%	1,457	7,965
New York	3.3%	8.0%	10,289	13,521
Ohio	5.4%	13.0%	10,633	7,816
Pennsylvania	9.5%	11.0%	16,863	11,972
Rhode Island	1.2%	11.0%	155	2,948
Vermont	11.3%	16.0%	1,111	2,459
West Virginia	13.6%	9.0%	3,939	4,005
Wisconsin	15.0%	10.0%	10,059	5,547
Total	5.5%	10.0%	83,821	122,200

[a] Ages 16 and older.

[b] Wildlife viewing away from home.

Source: U.S. Department of the Interior (DOI), Fish and Wildlife Service, and U.S. Department of Commerce, U.S. Census Bureau. 2007. *2006 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation*.

(2002) reported that more than 22% of households visiting Vermont in 2001 made the trip primarily for the purpose of viewing fall colors. Unfortunately, data on the total number or value of these trips are not available, although the high rates of participation suggest that numbers might be similar to the wildlife viewing estimates reported above.

Although these statistics provide useful indicators of the total recreational and aesthetic services derived from forests in the northeastern United States, they do not provide estimates of how these services are affected by terrestrial and forest acidification. Very few empirical studies have directly addressed this issue; however, there are two studies that have estimated values for protecting high-elevation spruce forests in the Southern Appalachians. Kramer, Holmes, and Haefele (2003) conducted a contingent valuation study estimating households' WTP for programs to protect remaining high-elevation spruce forests from damages associated with air pollution and insect infestation (Haefele, Kramer, and Holmes, 1991; Holmes and Kramer, 1995). The study collected data from 486 households using a mail survey of residents living within 500 miles of Asheville, North Carolina. The survey presented respondents with photographs representing three stages of forest decline and explained that, without forest protection programs, high-elevation spruce forests would all decline to worst conditions (with severe tree mortality). The survey then presented two potential forest protection programs, one of which would prevent further decline in forests along roads and trail corridors (one-third of the at-risk ecosystem) and the other would prevent decline in all at-risk forests. Both programs would be funded by tax payments going to a conservation fund. Median household WTP was estimated to be roughly $29 (in 2007 dollars) for the first program and $44 for the more extensive program.

Jenkins, Sullivan, and Amacher (2002) conducted a very similar study in 1995, using a mail survey of households in seven Southern Appalachian states. In this study, respondents were presented with one potential program, which would maintain forest conditions at initial (status quo) levels. It was explained that, without the program, forest conditions would decline to worst conditions (with 75% dead trees). In contrast to the previously described study, in this survey the *initial* level of forest condition was varied across respondent. In one version of the survey, the initial condition was described and shown as 5% dead trees, while the other version described and showed 30% dead trees. Household WTP was elicited from 232 respondents using a dichotomous choice and tax payment format. The overall mean annual WTP for the forest protection programs was $208 (in 2007 dollars), which is considerably larger than the WTP estimates reported by Kramer, Holmes, and Haefele (2003). One possible reason for this difference is that respondents to the Jenkins, Sullivan, and Amacher (2002) survey, on average, lived much closer to the affected ecosystem. Multiplying the average WTP estimate from this study by the total number of households in the seven-state Appalachian region results in an aggregate annual value of $3.4 billion for avoiding a significant decline in the health of high-elevation spruce forests in the Southern Appalachian region.

3.1.3 Effects on Regulating Services

Forests in the northeastern United States also support and provide a wide variety of valuable regulating services, including soil stabilization and erosion control, water regulation, and climate regulation (Krieger, 2001). As terrestrial acidification contributes to root damages, reduced biomass growth, and tree mortality, all of these services are likely to be affected;

however, the magnitude of these impacts is currently very uncertain. Forest vegetation plays an important role in maintaining soils in order to reduce erosion, runoff, and sedimentation that can adversely impact surface waters. In addition to protecting the *quality* of water in this way, forests also help store and regulate the *quantity* and flows of water in watersheds. Finally, forests help regulate climate locally by trapping moisture and globally by sequestering carbon. The total value of these ecosystem services is very difficult to quantify in a meaningful way, as is the reduction in the value of these services associated with N+S deposition.

3.2 Methodology for Estimating the Benefits of Reduced Terrestrial Acidification

Building on the framework developed in the REA, the methodology proposed in this section for estimating the benefits of reduced terrestrial acidification focuses on the value of increased provisioning services from increased sugar maple and red spruce timber harvests. This approach involves six main steps. First, the temporal framework describing the assumed future time path of deposition under the regulatory baseline scenario and multiple alternative control strategies is presented. The second step describes the estimated exposure–response models, which measure the empirical relationship between N+S deposition and growth in volume of live trees. The third step describes the simulation procedure, which applies the estimates from the exposure–response models to translate reduced deposition levels to changes in growth levels in the range of these tree species. The fourth step describes the method used to convert changes in growth levels for each plot to growth factors. In the fifth step, the method used to aggregate the predicted plot-level growth factors to regional-level growth factors is presented. This is followed by a description of alternative forest market models that can be used to estimate expected future market impacts and human welfare effects resulting from applying these growth factors. Each of these steps is described below in detail.

3.2.1 Temporal Framework

Figure 3-4 expands on the general temporal framework shown in Figure 1-2 to specifically represent the effects of alternative reductions in N+S deposition on forest productivity.

For a representative forest plot *i,* the current deposition level is depicted by D_i^0 . In practice, the 2002 CMAQ N+S deposition estimate corresponding to each plot will be used to represent D_i^0 .

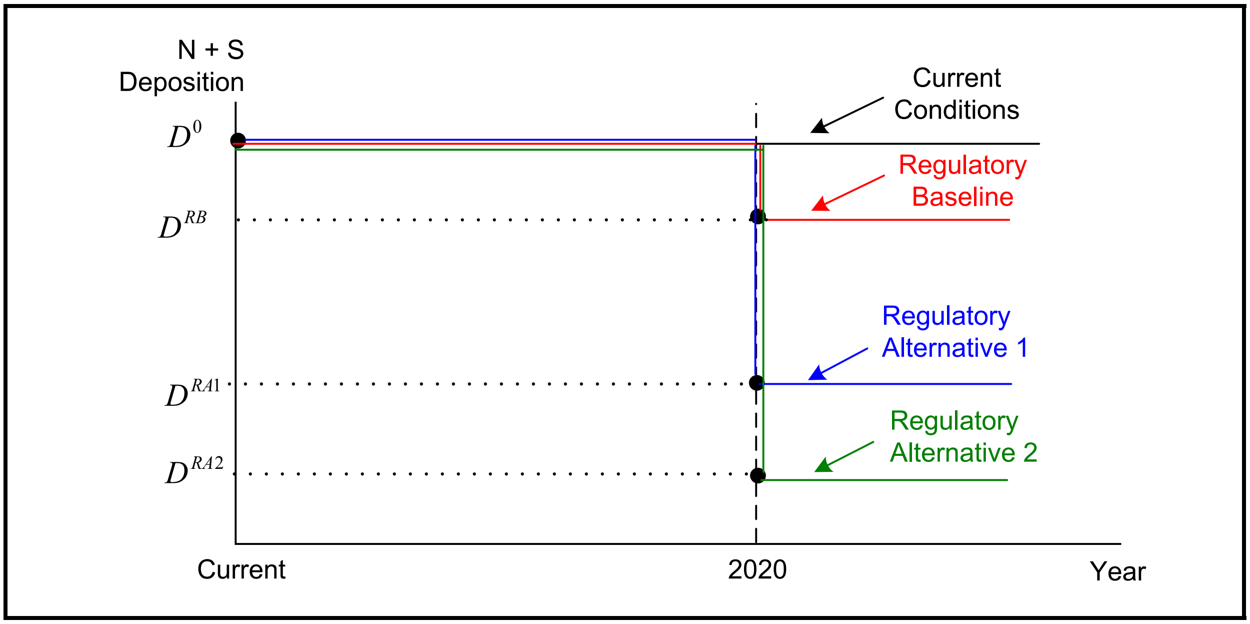

Figure 3-4. Temporal Framework: Expected Time Paths of Deposition Under Alternative Regulatory Scenarios

For the regulatory baseline (RB) scenario, we assume that deposition falls in 2020 from D_i^0 to D_i^{RB} and remains at that lower level for all subsequent periods.[3] For the regulatory alternative scenarios (RA1 and RA2), we also assume that deposition falls in 2020. In these cases, it decreases from D_i^0 to D_i^{RA1} and D_i^{RA2} respectively and also remains at these lower levels for all subsequent periods. In practice, 2020 CMAQ deposition estimates corresponding to each plot will be used to represent D_i^{RB}, D_i^{RA1} and D_i^{RA2}.

3.2.2 Exposure–Response Model

The next crucial step is to establish the relationship between N+S deposition and tree growth. This relationship must account for the fact that N deposition in forest systems can have either positive or negative impacts on tree growth. The growth of many forests in North America is limited by N availability (Chapin et al., 1993; Killam, 1994; Miller, 1988), and N fertilization is often a key component of forest management (Allen, 2001). Therefore, in such N-limited systems, N deposition may stimulate tree growth. In contrast, N additions in some systems can sometimes be greater than what trees require and can negatively impact tree health and growth

[3]As discussed in Section 1, this is a simplifying assumption. In reality, deposition levels under the regulatory baseline would be represented by a downward-sloping line to represent a gradual decline rather than a one-period drop in 2020.

(Aber et al., 1995; McNulty et al., 2005). In the context of acidic deposition of N+S, the positive versus negative impact of deposition on tree growth may depend largely on whether the critical load is exceeded by the deposition level. A critical load is defined as “a quantitative estimate of ecosystem exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur, according to present knowledge” (McNulty et al., 2007). When the N+S deposition is greater than the critical load, tree vigor and growth may be reduced because of the negative impacts of soil acidification (Figure 3-5a). Conversely, if N+S deposition is less than the critical load, tree growth may be stimulated because of a fertilizing effect of N deposition (Figure 3-5b). We refer to these two effects as the “acidification” and “fertilization” sides of the relationship, respectively. The transition point between growth stimulation and impairment would occur when N+S deposition is equal to the critical load.

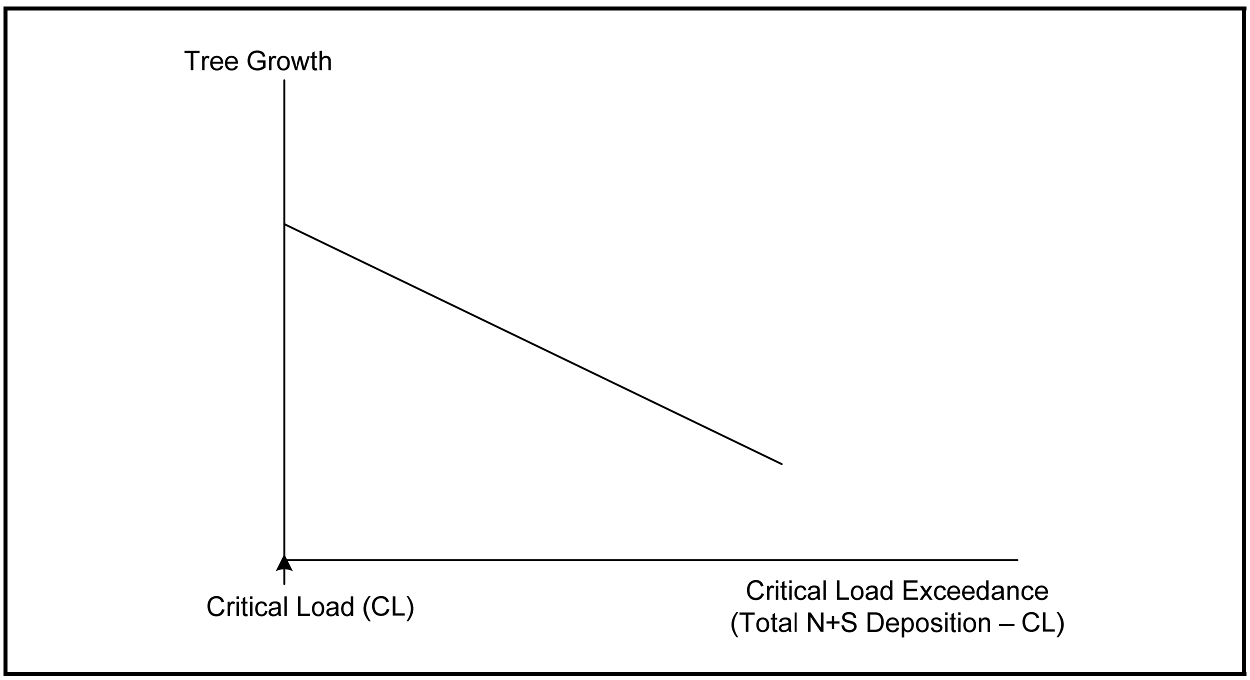

Figure 3-5a. Acidification Effect: Hypothesized Relationships between Tree Growth and Critical Load Exceedance

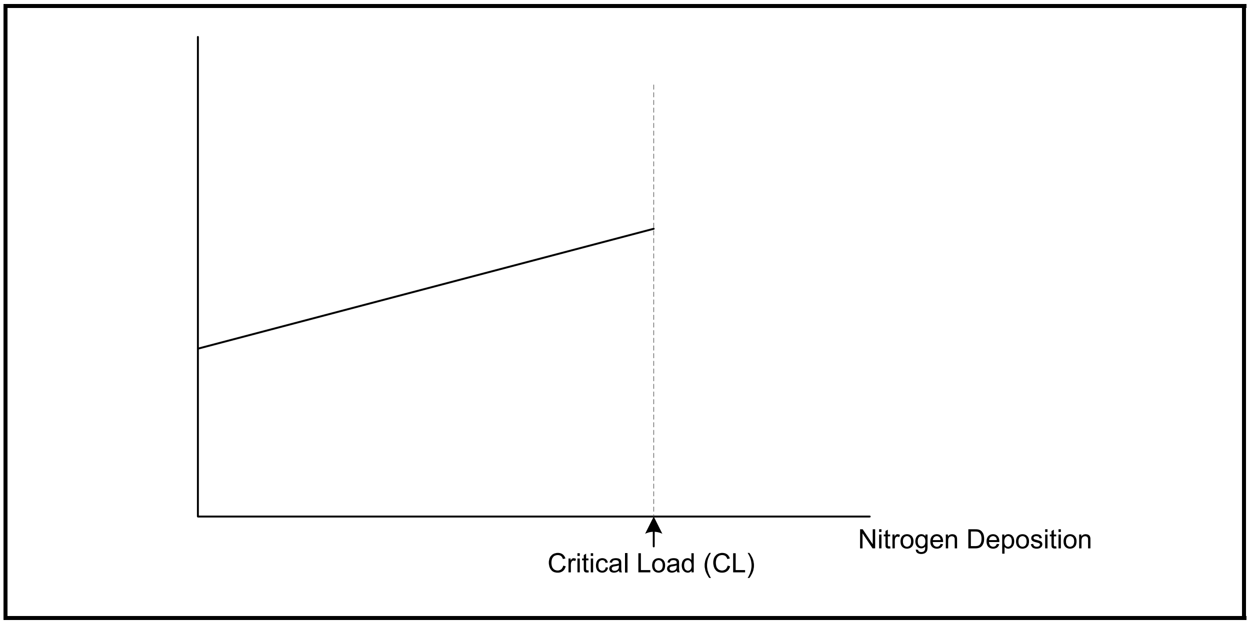

Figure 3-5b. Fertilization Effect: Hypothesized Relationship between Tree Growth and N Deposition Below the Critical Load

The acidification and fertilization effects were estimated separately using data from the USFS FIA for sugar maple and red spruce trees and the analyses described in greater detail in the REA report. The results of the exposure–response model measuring deposition exceedances above the most protective critical load (Bc/Al = 10.0) are summarized here.[4]

To estimate the acidification effect, the analysis used data from plots where 2002 N+S deposition exceeded the critical load (based on Bc/Al = 10.0 critical load estimates).[5] Data from the remaining plots with deposition below the critical load were used to estimate the fertilization effect. The key explanatory variable in the acidification regression is the critical load exceedance (CLE), and in the fertilization regression it is the total N deposition (ND) (both expressed as equivalents per hectare per year [eq/ha/year] at the plot). In both sets of analyses for both species, the other explanatory variables included linear and squared terms of average plot-level

[4]For details on the data used and alternative exposure–response models, see the case study report.

[5]For the acidification relationship, the analysis was restricted to plots north of the glaciation line. As detailed further in the REA case study report (Appendix 5, Attachment A-11 to A-13 and A-17), the statistical strength of the relationship between tree growth and critical load exceedance was better when the analyses were restricted to plots north of the glaciations line. This may be due to the weakness of the base cation weathering term used to estimate critical loads. The clay-substrate model was used to determine base cation weathering, and critical load experts have commented that this model may not be suitable for older, more weathered soils. Therefore, the estimates of critical loads and critical load exceedances for soils south of the glaciations line may be poorer and not show a relationship with tree growth.

tree volumes in cubic meters (m^3), and a dummy variable for state in which the plot is located. In general, the growth of a tree rises with age but at a decreasing rate.

Because data on the age were unavailable, average tree volume (which is highly correlated with tree age) was instead included as a proxy variable in the regression to control for this relationship. The purpose of the state variables is to control for other unobserved sources of variation in tree growth, which are related to a plot's general geographic location. Examples of potential unobserved factors include differences in data collection methods and measurements across reporting state, climatic factors, and geological characteristics.

The results of a multivariate ordinary least squares (OLS) regression for the acidification effects, using average plot-level tree growth (measured in cubic meters per year) as the dependent variable, are reported in Tables 3-2 and 3-3. The estimated value of the coefficient of the CLE (henceforth denoted by b_{acid}) was −3.344E-06 for sugar maple and was statistically significant at the .10% level (p-value of 0.101). The corresponding estimated value for red spruce was −5.162E-06, and this was statistically significant at the 5% level (p-value of 0.035).

The results of the linear regression analyses examining the fertilization relationship between nitrogen deposition and sugar maple and red spruce volume growth are presented in Tables 3-4 and 3-5, respectively.

For sugar maple, N deposition (henceforth denoted by b_{fert}) had a positive and significant (at the $p < 0.05$ level, $p = 0.00013$) impact on the above-ground volume growth of individual trees. These results suggest that the 2002 N additions, which ranged from 332 to 1,146 eq/ha/yr (4.7 to 16.1 kg N/ha/yr), acted as a fertilizer and stimulated growth on sites where N+S deposition did not exceed the critical load. In contrast to sugar maple, the red spruce in the tree growth–N deposition analyses displayed a negative, statistically insignificant relationship at the 5% level, $p = 0.2679$) between deposition and growth (Table 3-5).

3.2.3 Simulation Model for Plot-Level Tree Growth Changes

The purpose this step is to estimate, at a forest plot level, the effect of future reductions in deposition on forest tree growth. The deposition reductions of interest are those associated with both the regulatory baseline (RB) and the regulatory alternatives (RA), as described in Section 1. We provide an example of the simulation for 2020, the period of interest. The method would be analogous for any other year.

Table 3-2. Acidification Exposure–Response Model for Critical Load Exceedances and Sugar Maple Tree Growth: OLS Regression Results (for Plots above the Glaciation Line)

Explanatory Variables	Dependent Variable: Average Tree Growth (m^3/yr) Coefficient	t-statistic	p-value
Intercept	0.004875	1.48	0.1385
Critical load exceedance (CLE)	-3.344×10^{-6}	−1.64	0.1008
Average tree volume	0.021150	10.12	<.0001
Square of average tree volume	8.944×10^{-4}	1.1	0.27
Illinois	−0.001884	−0.31	0.755
Indiana	0.005452	1.63	0.1029
Iowa	−0.002052	−0.16	0.8743
Maine	−0.000895	−0.22	0.8245
Massachusetts	−0.008403	−1.82	0.0685
Michigan	0.000222	0.07	0.9456
Minnesota	0.000210	0.06	0.9553
Missouri	0.001850	0.35	0.7255
New Hampshire	−0.001647	−0.43	0.6696
New Jersey	0.001956	0.25	0.8042
New York	−0.000817	−0.25	0.8035
Ohio	−0.002104	−0.45	0.6522
Pennsylvania	−0.000803	−0.23	0.8177
Vermont	−0.005168	−1.51	0.131
Wisconsin	−0.002195	−0.68	0.4958
Number of observations	**2,205**		
Adjusted R^2	**0.1722**		

First, each plot *i* must be assigned a deposition level in 2020 under the regulatory baseline and the regulatory alternative scenarios (i.e., D_i^{RB}, D_i^{RA1}, and D_i^{RA2}). To associate individual plots with predicted future deposition levels, CMAQ grid-level estimates can be averaged at a county level and then applied to all plots located within each county. A more direct association between CMAQ grids and forest plots will most likely not be feasible because of confidentiality restrictions in the FIA data, which limit the amount of detail made available on specific plot locations.

Second, the scenario-specific critical load exceedances for each plot can be specified as $CLE_i^{RB} = D_i^{RB} - CL_i$, $CLE_i^{RA1} = D_i^{RA1} - CL_i$, and $CLE_i^{RA2} = D_i^{RA2} - CL_i$, respectively.

Table 3-3. Acidification Exposure–Response Model for Critical Load Exceedances and Red Spruce Tree Growth: OLS Regression Results (for Plots above the Glaciation Line)

	Dependent Variable: Average Tree Growth (m^3/yr)		
Explanatory Variables	**Coefficient**	**t-statistic**	**p-value**
Intercept	0.006034	4.96	<.0001
Critical load exceedance (CLE)	-5.162×10^{-6}	−2.12	0.0354
Tree volume	0.005590	1.26	0.2093
Square of tree volume	5.100×10^{-3}	1.23	0.2218
Maine	0.000285	0.32	0.7489
Massachusetts	−0.000132	−0.05	0.9629
New Hampshire	0.000435	0.42	0.6736
New York	−0.001805	−1.32	0.1897
Number of observations	**187**		
Adjusted R^2	**0.1963**		

Next, the effect of changes in deposition on tree growth under the RB and RA scenarios can be estimated using the exposure–response coefficients presented above. Estimating the net effect of reduced deposition on tree growth requires consideration of both the incremental effects due to reduced acidification and the detrimental effects due to reduced fertilization. There are three different cases, depending on whether the current, baseline, and alternative depositions in plots are higher or lower than the critical load.

In the following sections, to simplify the notation, we represent the regulatory scenarios—both the baseline (RB) and the alternative (RA)—by the superscript R and the current condition by 0.

Case 1: For plots where deposition is higher than the critical load under both the current and the regulatory scenarios (i.e., $CLE_i^0 > 0$ and $CLE_i^R > 0$), the following relationship can be used to estimate the increase in growth for a reduced exceedance level:

$$g_i^R = g_i^0 + b_{acid} * (CLE_i^R - CLE_i^0) \tag{3.1}$$

where

g_i^R = annual tree volume growth on plot *i* under regulatory scenario *R* (in m^3/yr)

Table 3-4. Fertilization Exposure–Response Model for N Deposition and Sugar Maple Tree Growth: OLS Regression Results (for Plots Below Critical Load)

	Dependent Variable: Average Tree Growth (m^3/yr)		
Explanatory Variables	**Coefficient**	**t-statistic**	**p-value**
Intercept	−0.00278	−0.53	0.5965
Nitrogen deposition	1.22×10^{-5}	3.22	0.0013
Average tree volume	0.01296	6.44	<0.0001
Square of average tree volume	-8.32×10^{-4}	−1.39	0.1645
Alabama	−0.00242	−0.39	0.6941
Illinois	−0.00131	−0.21	0.8350
Indiana	0.00338	0.65	0.5139
Iowa	5.19×10^{-4}	0.08	0.9392
Kentucky	6.37×10^{-4}	0.07	0.9479
Maine	0.00249	0.52	0.6034
Michigan	−0.00238	−0.51	0.6134
Minnesota	-4.20×10^{-4}	−0.09	0.9286
Missouri	−0.00167	−0.34	0.7302
New Hampshire	-3.34×10^{-6}	0.00	0.9995
New York	−0.00190	−0.35	0.7301
North Carolina	−0.01658	−2.17	0.0299
Ohio	−0.00535	−0.41	0.6812
Pennsylvania	0.00790	1.22	0.2241
Tennessee	5.46×10^{-4}	0.12	0.9080
Vermont	-7.88×10^{-4}	−0.08	0.9356
Virginia	0.00360	0.72	0.4693
West Virginia	0.00468	0.87	0.3847
Wisconsin	6.76×10^{-4}	0.14	0.8858
Number of observations		1,059	
Adjusted R^2		0.1175	

Table 3-5. Fertilization Exposure–Response Model for N Deposition and Red Spruce Tree Growth: OLS Regression Results (for Plots Below Critical Load)

	Dependent Variable: Average Tree Growth (m^3/yr)		
Explanatory Variables	**Coefficient**	**t-statistic**	**p-value**
Intercept	0.00704	1.57	0.1166
Nitrogen Deposition (ND)	-6.69×10^{-6}	−1.12	0.2679
Tree Volume	0.00955	3.72	0.0002
Square of Tree Volume	−0.00528	−2.31	0.0215
Maine	0.00121	0.44	0.6626
New Hampshire	4.5×10^{-4}	0.15	0.8825
Number of Observations		419	
Adjusted R^2		0.0351	

g_i^0 = annual tree volume growth on plot i under current conditions (in m^3/yr), based on the plot-level observations reported in the FIA data and used in the exposure–response modeling

b_{acid} = regression coefficient (slope) for critical load exceedance relationship (from Tables 3-4 and 3-5, equals -3.344×10^{-6} for sugar maple and -5.162×10^{-6} for red spruce)

Case 2: For plots where deposition is lower than the critical load under both the current and regulatory scenarios (i.e., $CLE_i^0 < 0$ and $CLE_i^R < 0$), the following relationship can be used to estimate the decrease in growth for a reduced deposition level:

$$g_i^R = g_i^0 + b_{fert}(ND_i^R - ND_i^0) \tag{3.2}$$

where

b_{fert} = regression coefficient (slope) for the deposition relationship (from Tables 3-4 and 3-5, equals 1.22×10^{-5} for sugar maple and for red spruce, we set this coefficient to zero since the coefficient is not significantly different from zero at the 5% level)

Case 3: For plots where deposition is higher than the critical load under the current and lower than the critical load under the regulatory scenarios (i.e., $CLE_i^0 > 0$ and $CLE_i^R < 0$), a combined approach is needed. That is, Equation (3.1) can be used to estimate the increased growth due to effects of eliminating exceedances (setting CLE_i^R =0) and Equation (3.2) to estimate the detrimental effects due to loss in fertilization on growth (setting ND_i^0 such that D_i^0 = CL). The difference in these estimated growths will be the net change in growth for the reduced deposition in the plot. It is possible that for some plots, the detrimental effect dominates the incremental effect; thus, the net change in growth is negative.

With this framework and the three cases outlined above, it is possible to look at both the separate and the combined effects of changes in acidification and fertilization. For example, to focus only on how reductions in acidification alone increase tree growth, the analysis can be done setting $b_{fert} = 0$.

It is also important to note that this simulation method assumes that growth adjusts instantaneously in response to the changes in deposition. In reality, there is likely to be a lag between the two. However, since the exposure–response model does not incorporate any dynamics, the timing of the adjustment is unknown.

3.2.4 Generating Plot-Level Growth Adjustment Factors.

For each plot, the simulated changes in tree growth associated with each regulatory scenario can then be translated into a corresponding growth adjustment factor as follows:

$$f_i^R = \frac{g_i^R}{g_i^0} \tag{3.3}$$

In other words, the growth adjustment factor *f* for scenario *R* represents the rate of increase (plus 1) in the tree growth for scenario *R*, relative to current growth.

3.2.5 Translating Plot Level Growth Adjustment Factors to Regional Averages

To apply the plot-level growth adjustment factors to a market model, an average growth factor for each region of the market model must be calculated. First the plot-level estimates of growth factor obtained in the previous step can be averaged for each county. Next, FIA data on county-level stocks of each species can be used as weights to calculate average growth factors for each region. The assumption underlying this weighting is that relative stocks are unchanged over time.

3.2.6 Forest Market Modeling to Estimate Market Effects/Welfare Changes Due to Changes in Tree Volume

This step describes the final link between altered N+S deposition levels and the changes in forest provisioning services, modeling the effect of applying a growth factor on public welfare. We expect to the FASOMGHG model described below to be the main modeling option for this analysis, but we also describe alternative approaches that can potentially be used to obtain these valuation estimates.

3.2.6.1 FASOMGHG

The FASOMGHG (**F**orest and **A**gricultural **S**ector **O**ptimization **M**odel—**G**reen **H**ouse **G**as version) can be used to calculate the resulting market-based welfare effects in the forest and agricultural sectors of the United States. Data obtained from the FIA will be used as inputs into FASOMGHG, which will enable the adaptation of the model for this application. The different components of these input data are described below.

FASOMGHG is a price-endogenous, dynamic, nonlinear programming model of the forest and agricultural sectors in the United States (Adams et al., 2005). The model simulates the allocation of land over time to competing activities in these two sectors and the resultant consequences for the commodity markets supplied by these lands. It was developed to evaluate the welfare and market impacts of public policies that cause changes in land use and activities both between and within the two sectors. FASOMGHG incorporates multiple market levels in the forest sector (logs, intermediate products, and final products) in the form of a manufacturing sector that transforms logs and their intermediate products into final products (Adams et al., 2005). Final products include solid wood products (e.g., softwood and hardwood lumber, softwood plywood) and fiber products (e.g., market pulp, recycled fiber, newsprint, linerboard).[6] The results from this model yield a dynamic simulation of prices, production, management, consumption, greenhouse gas (GHG) effects, and other environmental and economic indicators within these two sectors. For this application, FASOMGHG's key outputs include economic welfare measures, such as changes in producer and consumer surplus.[7]

The following discussion summarizes the main features of FASOMGHG and describes how they can be used and adapted for this application:

[6]For a complete listing of market products represented by FASOMGHG, please see Table 7-9 in Adams et al. (2005).

[7]For a detailed documentation of FASOMGHG, please see Adams et al. (2005).

- **Temporal Frame:** The time frame of this model is typically 70 to 100 years, and the model is solved on a 5-year time-step basis and the results of the model are presented at every time step. The base year for this model is 2002.
  - To apply the growth factors obtained in the previous step to FASOMGHG, annual growth factors must be translated to 5-year growth factors. These 5-year growth factors can then be applied beginning in 2020 and in all subsequent periods.
- **Geographical Regions:** FASOMGHG models forest and agricultural activity across the conterminous United States, which is broken into 11 market regions. Forestry production occurs in nine of these regions.
  - The selection of FASOMGHG regions for this model application can be determined by comparing the regions where sugar maples and red spruces grow with lists of states that make up the FASOMGHG regions.
- **Types of Forests:** Two types of forests are considered when evaluating policy effects in FASOMGHG—softwood and hardwood.
  - To apply FASOMGHG for this analysis, the main input required for the model is the percentage increase in *total hardwood* and *total softwood* growth rates by region for each 5-year time step in FASOMGHG. To address this requirement, the estimate of the average growth factor for *sugar maple tree* growth rate (obtained from the previous step) must be multiplied by the proportion of hardwoods in sugar maple production (shown in Table 3-6) for each FASOMGHG region, which ranges from 11% to 13%. Similarly, the estimate of the average growth factor in *red spruce tree* growth rate must be multiplied by the proportion of softwoods in red spruce production (also shown in Table 3-6) for the NE region.

Table 3-6. Proportions of Hardwood in Sugar Maple Production and Proportions of Softwood in Red Spruce Production, by FASOMGHG Region

	FASOMGHG Regions	Proportion of Hardwood/Softwood
Sugar Maple	NE	13%
	LS CB	11%
Red Spruce	NE	14.5%[8]

NE = Northeast: Connecticut, Delaware, Maine, Maryland, Massachusetts, New Hampshire, New Jersey, New York, Pennsylvania, Rhode Island, Vermont, West Virginia
LS = Lake States: Michigan, Minnesota, Wisconsin
CB = Corn Belt: All regions in Illinois, Indiana, Iowa, Missouri, Ohio
Source: U.S. Department of Agriculture, Forest Service, 2002.

[8]The RPA Assessment tables report the proportion of the spruce and balsam fir category as 29%. We assume that half of this is due to red spruce.

- **Forestland:** The FASOMGHG model does not track land under forest cover that produces less than 0.57 m^3/yr (called unproductive forestland) or on timberland that is reserved for other uses, because these are not part of the U.S. timber base. Endogenous land use modeling is only done for privately held land,[9] not publicly owned or managed timberlands. The model assumes that the amount of public land in forests does not adjust to market conditions but is set by the government. The proportions of the timberland under private and public ownership are shown in Table 3-7 (obtained from FIA data).

  – The average growth factors are applied to only forests growing on private land.

- **Welfare Measure:** Mathematically, FASOMGHG solves an objective function to maximize net market surplus. This is represented by the area under the product demand function (an aggregate measure of consumer welfare) less the area under the factor supply curves (an aggregate measure of producer costs). The value of the resultant objective function is consumer and producer surplus.

  – The welfare effects of applying the growth factor in 2020 will be obtained from FASOMGHG as the difference in annual net market surplus between each of the regulatory scenarios and the current scenario.[10] This will be obtained for every 5-year time step after 2020. The net present value of this stream of welfare effects will be calculated and discounted back to 2020 (using discount rates of 3% and 7%) and then annualized.

Table 3-7. Proportion of Timberland under Private and Public Ownership by FIA Region[a]: 2002

FIA Region	Private Timberland	Public Timberland
Northeast	87%	13%
NorthCentral	28%	72%

[a] The states in the Northeast FIA region correspond exactly to states in NE in FASOMGHG. The states in the NorthCentral FIA region correspond exactly to states in LS and CB in FASOMGHG.

Source: U.S. Department of Agriculture, Forest Service, 2002, Table 10.

FASOMGHG offers several advantages. Most importantly, it provides an established, sophisticated, and data-rich model for estimating the value (i.e., increase in consumer and producer surplus) of expected future increases in timber harvests and sales in the forest and agricultural sectors of the United States (Adams et al., 2005). A potential disadvantage of FASOMGHG for this application is that, because of the large geographic and economic scope of

[9]FASOMGHG assumes that private landowners behave as profit maximizers. However, that may not always be true because they may be utility-maximizers that manage based on recreational opportunities, etc., in addition to timber value.

[10]The current scenario can be represented by running FASOMGHG with its existing tree growth rate coefficients (i.e., with no growth adjustment factors).

the model, it is not necessarily the most appropriate tool for examining relatively small changes in yields for only two commercial tree species in one to two regions of the country. In addition, the model involves considerable runtime and, thus, can be resource intensive.

To address the potential limitations of FASOMGHG, we believe it may also be useful to consider two simpler modeling approaches, which are described briefly below. Some of the limitations of these approaches are also noted.

3.2.6.2 Partial Equilibrium Market Modeling Approach

In this approach, a demand and supply framework can be developed and used to calculate the welfare effects of altered tree growth factors due to reduced deposition to the timber producers and consumers in 2020. Holmes (1992) describes a similar approach to estimate welfare effects for a decline in southern pine forest productivity in the United States. Profit maximization by timber producers yields an upward-sloping supply curve of timber volume. Figure 3-6 provides an example of this framework where the incremental effects due to reduced acidification dominate the detrimental effects resulting from reduced fertilization. In this case, reduced deposition in 2020 would translate to a net increase in growth. Applying the growth factors obtained in the previous step would, consequently, translate to an increase in the volume of trees in the same period. This would result in a rightward shift of the supply curve. The supply curves corresponding to the current, RB, and RA deposition levels, are depicted by $S^{CURRENT}$, S^{RB}, and S^{RA1}, respectively. Welfare effects of reduced deposition from the RB level to the RA1 level can be estimated by measuring the change in consumer and producer surplus from this model (area $AE^{RB}E^{RA1}$), assuming a downward-sloping demand curve for timber (D).

Implementing this model requires data. Baseline quantity and price data would be obtained from FIA and stumpage price reports.[11] Estimates of demand and supply elasticities will be obtained from the literature.[12] Estimates of the growth factors obtained from the previous step would be used to calculate the change in volume and this would be used to parameterize the rightward shift in the supply curve.

<u>Limitations of using this approach:</u> This model is a static one as opposed to the dynamic framework used in FASOMGHG. This can result, for example, in "short-sightedness" of producers who may tend to overharvest if they do not associate any value with the stock of trees at the end of a finite time period. Also, this model does not include any interactions with the

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[11]These are available at http://www.srs.fs.usda.gov/econ/data/prices/index_t.htm.
[12]One potential study that can be used to obtain elasticities is that by Luppold (1984).

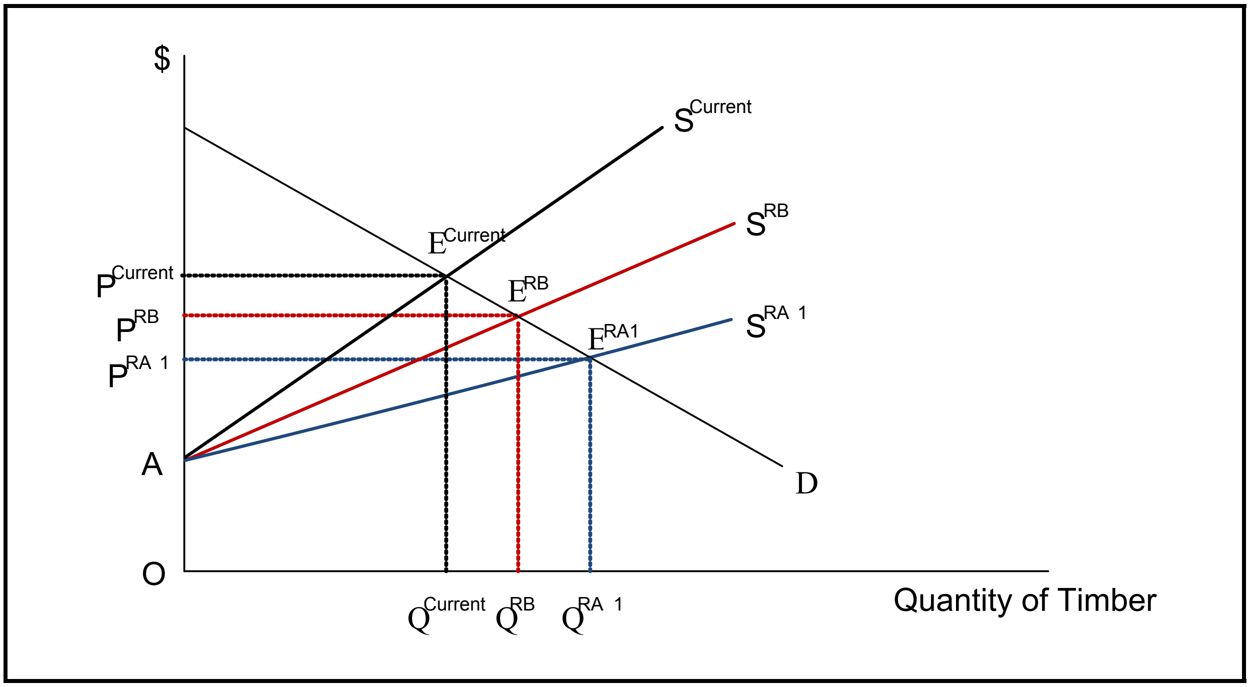

Figure 3-6. Partial Equilibrium Effects of Increased Tree Growth

agricultural sector and, thus, cannot account for shifts between the two sectors. The results from the model would, however, be transparent and easy to interpret.

3.2.6.3 A Forest Inventory Modeling Approach

In this approach, applying the growth factor results in changes in tree volumes and, consequently, results in an increased value of timber inventories. A similar method was suggested as an approach to quantify economic impacts of changes in timber growth in a case study of the Adirondacks (EPA, 2007). The inventory of timber from trees in a representative region at the current period can be described by the following expression:

$$\text{Inventory in 2020} = f(\text{inventory from previous year} + \text{net growth in 2020} - \text{harvest})$$

where net growth is defined as the change in volume from the previous year and is obtained as a product of the growth rate under the regulatory baseline and tree volume in the previous year.

In this application, estimates of the growth factor obtained from the previous step would be multiplied by the growth rate under the regulatory baseline to obtain a net growth in 2020 under the RA. Thus, we would obtain estimates of the inventory under RB (using growth levels corresponding to RB) as well as estimates of the inventory under the RA (using growth levels corresponding to the RA). The market effects of altered deposition levels can be estimated as the value of the change in the timber inventory and this is given as follows:

Change in the Value of Timber in 2020 = ([inventory under RA – inventory under RB] * stumpage price)

Reduced deposition, for example, will result in a higher net growth rate and higher volume and, thus, an increased value of timber inventory, given prices.

Implementing this model requires information on existing harvests and inventories, which can be obtained from the FIA data. In addition, stumpage price data can be obtained from stumpage price reports.[13]

<u>Limitations of using this approach:</u> This approach is not an optimization framework, and it does not model any changes in producer (or consumer) behavior. Prices are treated as completely exogenous in this model. Moreover, it does not directly yield estimates of economic welfare such as producer and consumer surplus. Rather, it provides an approximation of benefits. Although the somewhat restrictive assumptions and structure of this approach may be appropriate for a case study over a small area such as the Adirondacks, they are likely to be less appropriate for applying to a wider regional area. Nevertheless, in comparison to FASOMGHG, the model is less resource intensive. It also provides results that are transparent and easy to interpret.

3.2.7 Summary of Modeling Framework

Figure 3-7 summarizes the key modeling steps for assessing the benefits of reduced terrestrial acidification. For the step describing the benefits analysis, we have presented the FASOMGHG model since we are currently using this. However, if any of the alternative market models described above are used instead of FASOMGHG, they could be incorporated in this step as well.

3.3 Options for Expanding the Analysis of Forest Acidification Effects

The approach described above for estimating the benefits of reduced terrestrial acidification offers several advantages, including a significant geographic scope. It incorporates most of the Midwestern and Northeastern states with both high deposition and significant numbers of the affected tree species. Nevertheless, the analysis could be usefully expanded or

[13]These are available at http://www.srs.fs.usda.gov/econ/data/prices/index_t.htm.

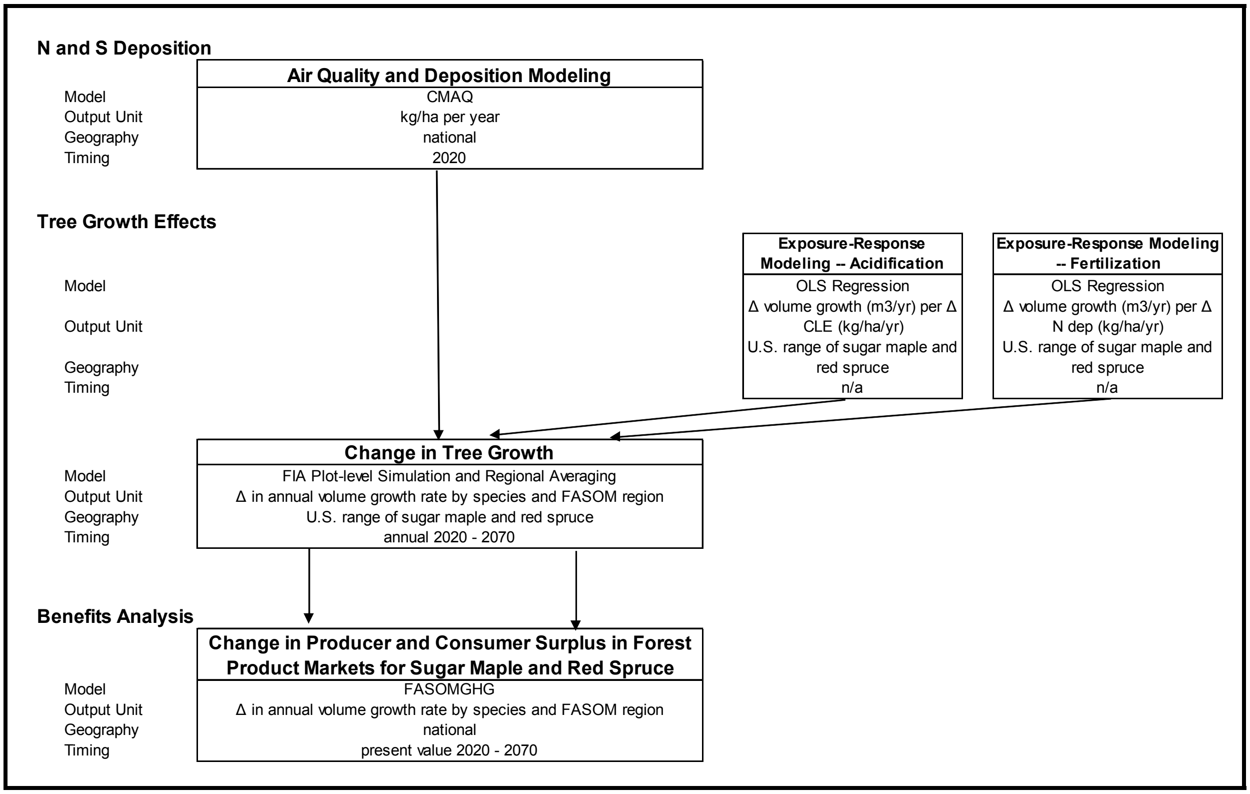

Figure 3-7. Key Modeling Steps for Assessing the Benefits of Reduced Terrestrial Acidification

modified in a number of ways. The following discussion describes and prioritizes a number of options for expanding the analysis.

3.3.1 Exposure–Response Modeling

The analyses assessing the relationships between tree growth and critical load exceedances or N deposition could be improved by modifying the current data sets and by adding other relevant data and supporting analyses. Through such modifications and additions, improving the statistical strength of the relationships between tree growth and acidifying N+S deposition may be possible. The current analysis finds effects of the expected sign and reasonable magnitude, but the statistical significance of these effects is somewhat weak. In addition, it may possible to broaden the application of the trends across a larger landscape and variety of tree species.

3.3.1.1 Analysis Expansions with the Current Dataset

Within the current data sets used to compare sugar maple and red spruce growth and critical load exceedances or N deposition, several potentially influential components were not included in the original analyses. Only live sugar maple trees were assessed in the regression analyses. However, studies have established that acidifying N+S deposition can result in tree mortality (DeHayes et al., 1999; Drohan and Sharpe, 1997; Perkins et al., 2000; Schaberg et al., 2000). Therefore, the critical load exceedance—tree growth regressions may be improved by including dead trees in the analyses. In the FIA database, when a tree is recorded as dead for the first time, the total volume of that tree is considered negative volume growth over the most recent measurement period.[14] During the original analyses, there was uncertainty regarding the validity of including the dead tree negative-volume growth values (as calculated in the FIA database) in the analyses. However, the potential importance of these trees in the tree growth–critical load exceedance relationship may warrant renewed consideration of including these trees in future analyses.

A second alteration to the current data set that could be incorporated into regression analyses is the removal of "ingress" trees from the data set. According to USFS FIA database methodology (C. Alerich, personal communication), trees must be at least 12.7 cm in diameter to be registered as a tree within the database. When a tree reaches this size, the full volume of the stem is incorporated into the volume growth measurement, and, in many cases, these first-time measurements would be larger than the actual annual growth rate of the tree. Therefore, the removal of such trees with incorrectly high volume growth values from the analyses could improve the estimates and statistical significance of the relationship between tree growth and critical load exceedance or N deposition.

3.3.1.2 Analysis Expansions with Additional Data

Regression analyses comparing tree growth and critical load exceedances or N deposition could also be improved with the addition of other relevant data and variables to the current data sets. In the original analyses, only state categorical variables and current tree volume were included as covariates to account for the influences of location-specific factors (e.g., climate, water availability) and tree size on sugar maple and red spruce growth. However, additional factors may also influence the relationship between growth and critical load exceedance or N deposition, and these factors could be included in future regression analyses. For example, incorporating total stand basal area could be included to account for the influences of stand density on tree growth. Elevation data could also be used in the analyses to better account for the

[14] Dead tree volume growth is calculated as a difference in volumes ($v_2 - v_1$) divided by the time between sequential measurement period ($t_2 - t_1$). When a tree is recorded as dead, it is assigned a v_2 value of "0." Therefore, the associated volume growth is equal to the entire tree volume divided by the difference in the number of years between the current and last measurement cycle (C. Alerich, personal communication).

effects of location and differences in growing season length on the growth of red spruce and sugar maple. Measurement year and time between measurements could help control for the influence of year-to-year variation in conditions and measurement methodology on the growth data. Climatic variation (e.g., rainfall, temperature) could be included to account for the influence of drought and frost conditions on tree growth (Driscoll et al., 2003; Johnson and Siccama, 1983; Schaberg et al., 2002; Sheppard, 1994). In addition, although current tree volume was included as a covariate in the original analyses, the FIA variable used to account for tree volume (VOLCFNET) may not have been the most suitable covariate for volume growth (FGROWCFAL). VOLCFNET is based on merchantable volume (e.g., pulp and sawlog), whereas FGROWCFAL is based on the growth of sound wood (USFS, 2008). The difference between the two measurements is cull wood that is sound but not merchantable because of circumstances such as location on the tree or tree branchiness. A more suitable tree volume variable would be VOLCFSND, which is based on the net volume of sound wood, and the use of this variable as the covariate to tree growth may improve the regression analyses.

The regression analyses comparing tree growth and critical load exceedances or N deposition could also be potentially improved and enhanced with the addition of supporting analyses and data sets. As suggested by critical load experts and by the results of the critical load exceedance—tree growth regression analyses for sugar maple in the REA and ecosystem services analysis, the clay-substrate methodology used to estimate base cation weathering (BC_w) in the critical acid load calculations may not provide good estimates of BC_w in older, more weathered soils south of the southern extent (glaciation line) of the most recent glaciation (~20,000 years before present [ybp]) (Figure 3-8). Although the clay-substrate method is one of the most commonly used methods to estimate BC_w in North America (McNulty et al., 2007; Ouimet et al., 2006; Pardo and Duarte, 2007; Watmough et al., 2006), other models, such as PROFILE (Sverdrup and Warfvinge, 1993a), which are based on soil mineralogy, may provide better estimates of the contribution of base cations from soil weathering. It is, therefore, possible that critical acid load estimates and analyses of the relationship between tree growth and critical load exceedance may be improved by using a more suitable method to estimate BC_w. This improvement would be particularly noticeable in locations south of the glaciation line.

The regression analyses and the extent to which the tree growth and critical load exceedance or N deposition relationships could be applied to a larger geographic area could also be improved by including additional tree species in the evaluations. Laboratory studies have established that at least 31 of the native tree species found growing in North America are sensitive to Bc/Al soil solution ratios that can be created by acid deposition (Sverdrup and

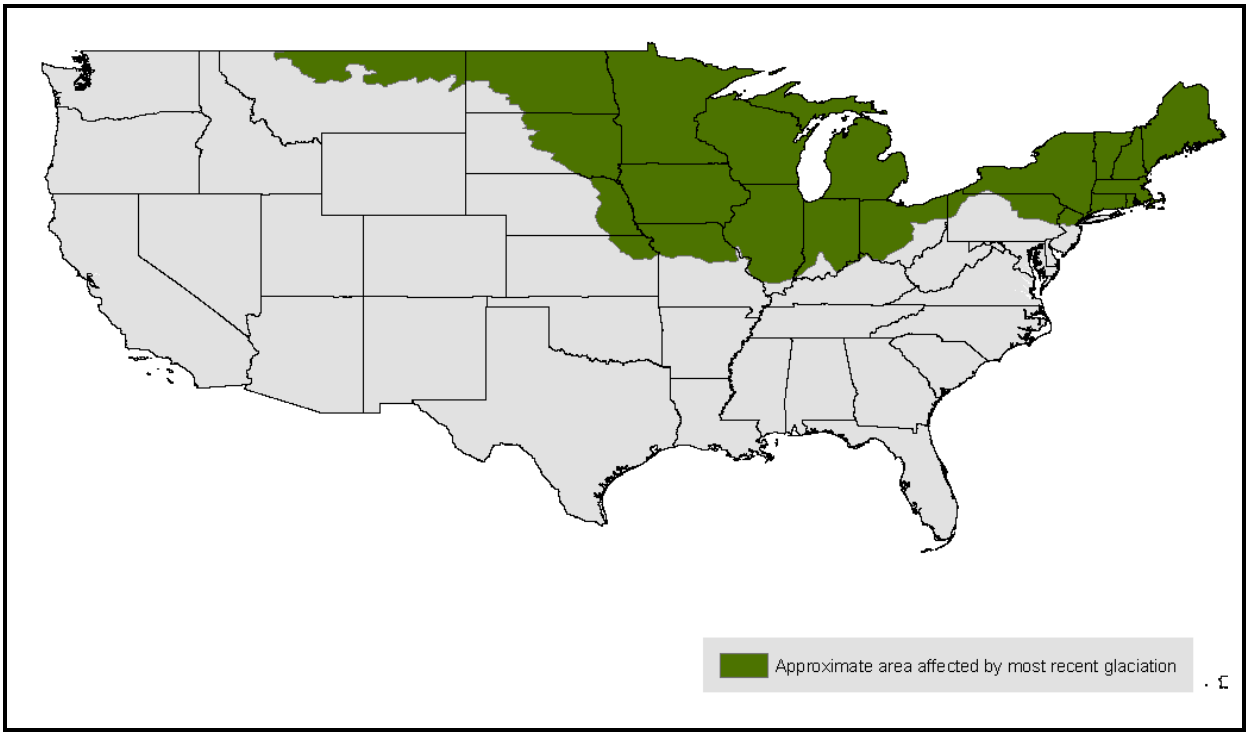

Figure 3-8. Areas of Continental United States that Were Covered during the Last Glacial Event (~20,000 ybp) (Reed and Bush, 2005)

Warfvinge, 1993b) (Table 3-8). Therefore, the critical load and regression analyses could be expanded to address multiple species and regions throughout the United States.

3.3.2 Recommended Prioritization of Analysis Expansions Options

As discussed above, a variety of modifications and additions could improve the analyses of the relationships between tree growth and critical load exceedance or N deposition. However, given the number of options and the constraints of limited resources, it may not be possible to incorporate all the recommended improvements. Therefore, here we present three analysis priorities that collectively achieve the objective of expanding the scope and breadth of the tree growth to critical load exceedance or tree growth to N deposition relationships, while maximizing the accuracy and strength of the analyses.

Priority 1—Increase Statistical Strength and Accuracy of Analyses

The first recommendation involves improving the current analyses and red spruce and sugar maple data sets by using the most appropriate terms, variables, and data. As described in Section 1.1.1, the inclusion of "ingress" trees, trees that entered the FIA database during the most recent measurement period, may represent data points with erroneously high volume growth

Table 3-8. North American Tree Species and Associated Soil Solution Base Cation to Aluminum (Bc/Al) Ratio at which Biomass Volume Is Reduced by 20% Relative to Controls (Sverdrup and Warfvinge, 1993b)

Tree Species			
Common Name	**Scientific Name**	**$Bc/Al_{(crit)}$**	**Commercial Uses[a]**
Western Red cedar	*Thuja plicata*	0.09	Valuable wood for wide variety of products
N. White Cedar	*Thuja occidentalis*	0.1	Rustic fencing, posts, cabin logs, umber, poles, shingles, other specialty products
Western Hemlock	*Tsuga heterophylla*	0.2	Lumber and pulp
Douglas Fir	*Pseudotsuga menziesii*	0.3	Lumber and pulp
Honey Locust	*Gleditsia triacanthos*	0.4	Local uses: fence posts, lumber, pallets, crating, and general construction
Sitka Spruce	*Picea sitchensis*	0.4	Lumber, pulp, and many specialty products
E. White Pine	*Pinus strobus*	0.5	Furniture, Christmas trees, and specialty products
Slash Pine	*Pinus elliottii*	0.5	Naval stores and lumber
White Spruce	*Picea glauca*	0.5	Pulp, lumber, and specialty products
American Beech	*Fagus grandifolia*	0.6	Flooring, furniture, veneer, and containers
N. Red Oak	*Quercus rubra*	0.6	Lumber
Pin Oak	*Quercus palustris*	0.6	Lumber
Sand Pine	*Pinus clausa*	0.6	Fuelwood, biomass, Christmas trees
Shortleaf Pine	*Pinus echinata*	0.6	Lumber, plywood, other structural materials, and pulp
Sugar Maple	*Acer saccharum*	0.6	Syrup, lumber, pulp, and firewood
Black Spruce	*Picea mariana*	0.8	Pulp, lumber, Christmas trees, and other products
Balsam Fir	*Abies balsamea*	1.1	Pulp, lumber, and Christmas trees
False Acacia (Black locust)	*Robinia pseudoacacia*	1.2	Not a commercial timber species
Fraser Fir	*Abies fraseri*	1.2	Christmas trees
Pitch Pine	*Pinus rigida*	1.2	Lumber for rough construction
Red Spruce	*Picea rubens*	1.2	Pulp, lumber, and specialty products
Scotch Pine	*Pinus sylvestris*	1.2	Pulp, lumber,and Christmas trees
Scrub/Virginia pine	*Pinus virginiana*	1.2	Pulp and lumber
Jack Pine	*Pinus banksiana*	1.5	Pulp, lumber, and round wood
Loblolly Pine	*Pinus taeda*	1.5	Lumber and pulp
Gray Birch	*Betula populifolia*	2	Specialty items
Longleaf Pine	*Pinus palustris*	2	Broad range of forest products
Paper Birch	*Betula papyrifera*	2	Veneer, pulp, and many specialty products
Ponderosa Pine	*Pinus ponderosa*	2	Lumber and pulp
Red Pine	*Pinus resinosa*	2	Lumber and pulp
Yellow Birch	*Betula alleghaniensis*	2	Lumber

[a]Commercial uses from USFS (2009a,b).

values. Therefore, as a first step, these "ingress" trees should be removed from the data sets used in the regression analyses. The RECONCILECD variable within the FIA database provides information on the "ingress" status of a tree (USFS, 2008). Secondly, the FIA tree volume variable (VOLCFNET) that was used as a covariate in the regression analyses should be replaced with VOLCFSND. This VOLCFSND variable is more consistent with the volume growth variable (FGROWCFAL) used in the REA analyses.

These two modifications to the data sets were selected as the first recommendation because, in addition to improving the strength and accuracy of the analyses, they would require comparatively little resources and time and involve changes that would be included in subsequent priority tasks and future analyses.

Priority 2—Improve Critical Load and Critical Load Exceedance Estimates

Approximately 26% of the sugar maple plots used in the REA tree growth—critical load exceedance regression analyses were located on older soils south of the most recent glacial advance. As previously discussed, the clay-substrate method to determine BC_w within the critical load calculations may not provide accurate estimates of weathering in these older, unglaciated soils. Therefore, to improve the accuracy of the tree growth–critical load exceedance analyses, our second recommendation is to calculate base cation weathering using a more appropriate method such as PROFILE. This would involve acquiring soil and mineralogy data to support PROFILE analyses in the geographical area of interest. Preliminary analyses of GIS databases and layers indicate that the necessary data are available but coverage throughout the United States may be incomplete.

This modification to the data sets was selected as the second recommendation because improved accuracy in the critical acid load and exceedance estimates would support more accurate analyses of the relationship between tree growth and critical load exceedance. Such an improvement would be particularly important for analyses in locations that were not glaciated during the last glacial period. A total of 38 states in the continental United Statescover an area that is south of the most recent glacial advance. Therefore, the use of a better BC_w model could benefit critical load analyses in the majority of the land area in the United States.

Priority 3—Expansion of Analyses to Include More Tree Species

The REA analysis of terrestrial acidification was restricted to two tree species and a total of 24 states in the northeastern and midwestern regions of the United States. Therefore, the third recommendation is to expand the tree growth and critical load exceedance or N deposition analyses to include more tree species. With such an expansion, the number of species evaluated within a given area could be increased and/or the geographical area considered in the analyses could be expanded. By increasing the number of tree species in a given area, it would be possible to gain a more complete understanding of the impacts of acidifying N+S deposition on the ecosystem services provided by a forest, and not just provisions supplied by one or two tree species. By increasing the geographical coverage of the analyses, it would be possible to examine the impact of acidifying N+S deposition on biological receptors or end points over a larger geographical area. As outlined in Section 1.1.2, at least 31 tree species found growing in the United States appear to have a sensitivity to Bc/Al soil solution ratios that could be caused by acid deposition (Table 3-8). Collectively, these species cover the western, midwestern, northeastern, and southern regions of the United States (Figure 3-9).

When selecting the tree species to include in the expansion analysis, there should be at least three main considerations. The first consideration is to give higher priority to commercially important species that produce the highest value of forest products. Such products include lumber, pulpwood, syrup, Christmas trees, and specialty products such as veneer, furniture, flooring, poles, siding, and shingles.

A second consideration is to give lower priority to tree species that are commonly fertilized with N on managed forest lands. N fertilization may mask any impact of acidifying N+S deposition on tree growth. For example, loblolly pine (*Pinus taeda*), slash pine (*Pinus elliotti*), and Douglas-fir (*Pseudotsuga menziesii*) are routinely fertilized with as much as 200 kg N/ha several times during a rotation (J. Rojas *pers. comm.;* T. Fox personal communication). To limited degrees, lodgepole pine (*Pinus contorta*), ponderosa pine (*Pinus ponderosa*), western hemlock (*Tsuga heterophylla*), and western redcedar (*Thuja plicata*) may also be fertilized with N, but rarely on a commercial scale.

A third consideration in the expansion of the regression analyses is the distribution of N+S deposition across the United States Areas of high deposition are located mainly in the northeastern and midwestern regions with moderate to high deposition levels in the south. There are only small pockets of high N+S deposition in the western U.S. states. To get a good indication of the relationships between tree growth and critical load exceedance or N deposition, analyses should be conducted in areas with a wide range of N+S deposition levels (that presumably have corresponding wide ranges in critical load exceedance values).

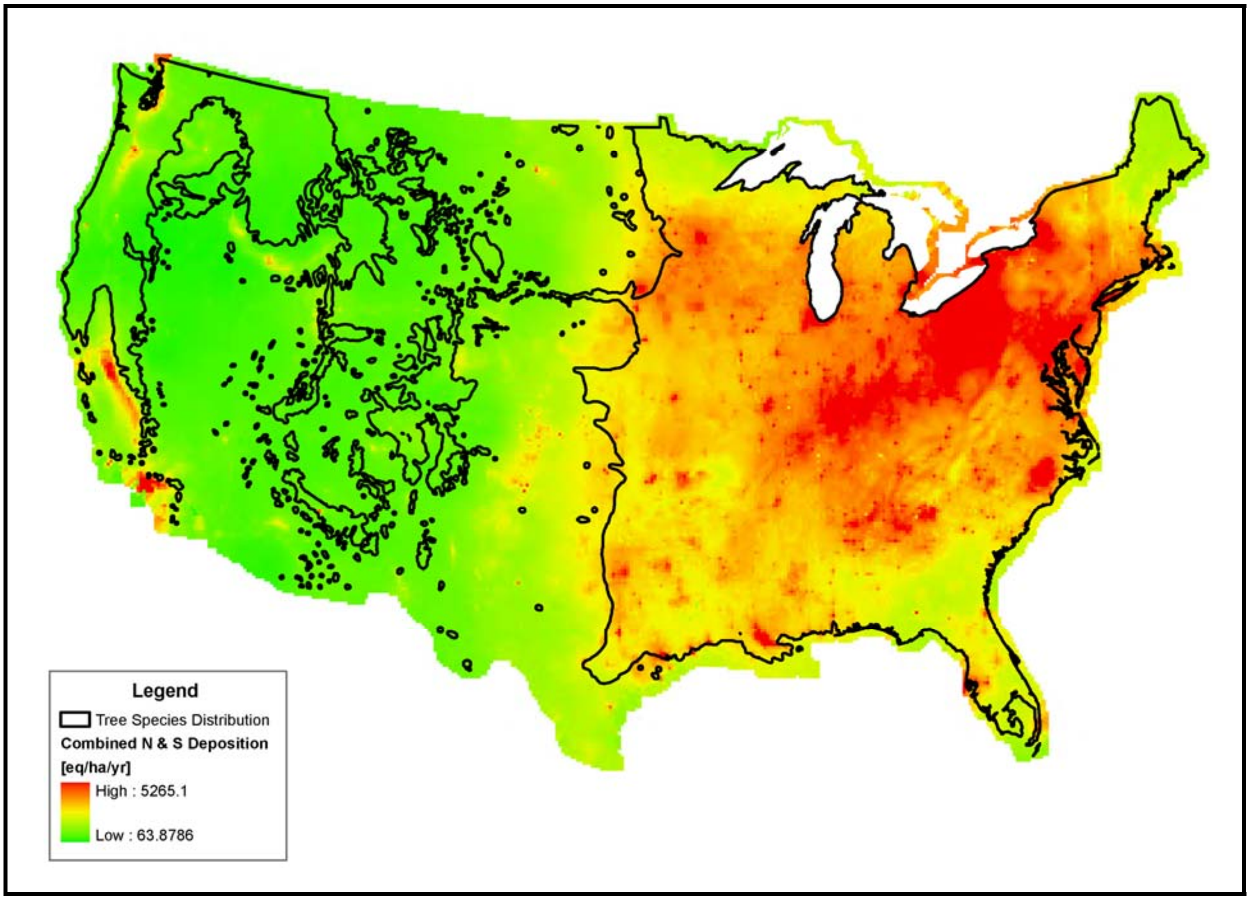

Figure 3-9. The Geographical Area of the Continental United States That Is Occupied by the 30 Tree Species That Have Shown a Sensitivity to Bc/Al Soil Solution Ratios That Could Be Caused by Acidifying N+S Deposition

Considering a combination of the three factors outlined above, we recommend that the initial expansion of the analyses be focused on the northeastern, Midwestern, and/or southern regions of the United States and be restricted to 16 species (Table 3-9). These species are found in three identified regions, are not routinely fertilized, and are considered commercially important. "Commercially important," for the purposes of this report, has been defined as a tree species that is used to produce commercial products such as pulp, lumber, flooring, furniture, veneer, Christmas trees, and specialty items, and is harvested at a rate of at least 1% of timber

Table 3-9. Total Standing Volume and Annual Removals of Growing Stock Trees (on FIA Database Forest Land or Timber Land) for Species Present in Multiple States within the Northeastern, Midwestern and/or Southern Regions of the United States

Tree Species	Volume of Removals (m^3)	Volume of Growing Stock on Timber Lands (m^3)	Volume of Growing Stock on Forest Land (m^3)	Removals Volume (% of Timberland or Forest Land Growing Stock)
Shortleaf Pine	16,119,916	335,295,164	340,091,876	4.7 – 4.8
Sugar Maple	8,137,160	685,168,208	735,222,075	1.1 – 1.2
N. Red Oak	7,861,601	617,759,055	644,949,944	1.2 – 1.3
E. White Pine	6,369,901	468,472,752	491,628,575	1.3 – 1.4
Scrub/Virginia Pine	6,330,946	142,193,707	145,398,438	4.4 – 4.5
Longleaf Pine	4,597,325	116,070,523	131,591,546	3.5 – 4.0
Red Spruce	3,219,219	41,809,134	163,482,880	2.0 – 7.7
American Beech	3,066,299	234,179,886	257,305,792	1.2 – 1.3
Balsam Fir	3,034,463	78,148,521	152,248,682	2.0 – 3.9
Paper Birch	2,304,335	138,107,792	161,936,357	1.4 – 1.7
Red Pine	1,802,691	184,923,798	154,971,213	1.0 – 1.2
Jack Pine	1,496,271	31,735,712	35,022,303	4.3 – 4.7
Yellow Birch	1,446,655	118,423,447	152,463,408	0.9 – 1.2
White Spruce	1,038,534	41,230,641	68,302,856	1.5 – 2.5
Black Spruce	945,267	40,854,383	61,229,035	1.5 – 2.3
Sand Pine	707,634	16,933,798	17,215,031	4.1 – 4.2

land[15] or forest land[16] growing stock each year. We are assuming that the vast majority of the volume removals[17] listed in Table 3-9 are due to harvesting rather than land-use change.[18]

[15]"Timber land" within the FIA database is defined as "Forest land that is producing or is capable of producing crops of industrial wood and not withdrawn from timber utilization by statute or administrative regulation. (Note: areas qualifying as timber land are capable of producing in excess of 20 cubic feet per acre per year of industrial wood in natural stands" (Smith et al., 2009).

[16]"Forest land" within the FIA database is defined as "Land at least 120 feet wide and 1 acre in size with at least 10 percent cover (or equivalent stocking) by live trees of any size, including land that formerly had such tree cover and that will be naturally or artificially regenerated. Forest land includes transition zones, such as areas between forest and nonforest lands that have at least 10 percent cover (or equivalent stocking) with live trees and forest areas adjacent to urban and built-up lands….Tree-covered areas in agricultural production settings, such as fruit orchards, or tree-covered areas in urban settings, such as city parks, are not considered forest land" (Smith et al., 2009).

[17]"Removal" within the FIA database is defined as the removal of trees from a plot due to harvesting or diversion of the plot from forest to nonforest condition (for some analyses, this may also include land diverted to reserved forest land or other forest land) (Bechtold and Patterson, 2005).

[18]Within the FIA database, it is not possible to distinguish between harvesting and land-use change as the source of removal of trees from a plot.

As discussed earlier, there are at least two main ways to expand the number of species analyzed with respect to forest acidification effects. One approach is to analyze several species jointly, by focusing on areas that fall within the range of multiple species. Figure 3-10 represents the recommended location and species mixture for this approach. The distributions of a large number of species overlap within the northeastern region of the U.S., and as many as 10 tree species could be present in the same plots within the outlined area in Figure 3-10. These 10 species are indicated with a checkmark in the second column of Table 3-10. This area also captures a range of N+S deposition levels, although it is dominated by lower amounts of deposition (366.1 to 1,166.4 eq/ha/yr). Therefore, this region would be a good location to conduct a more comprehensive examination of the impacts of acidifying N+S deposition on the services provided by forests or a mixture of tree species.

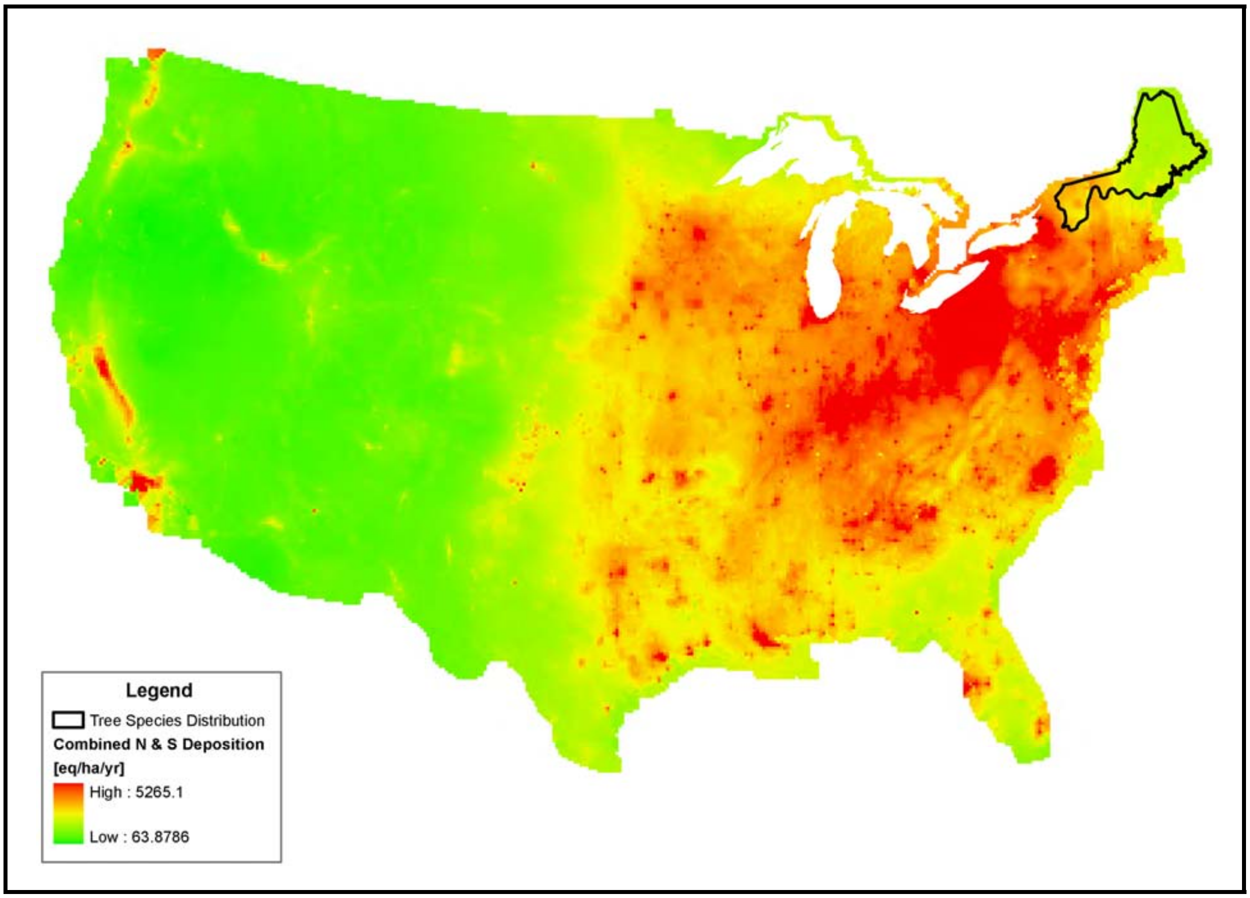

Figure 3-10. The Geographical Area of the Continental United States That Is Occupied by 10 of the 30 Tree Species with Overlapping Distributions

Table 3-10. High-Priority Tree Species for Expanded Analysis of Forest Acidification Effects

Tree Species	Combined Species Approach	Individual Species Approach
Shortleaf Pine		√
Sugar Maple	√	√
N. Red Oak	√	√
E. White Pine	√	√
Scrub/Virginia Pine		√
Longleaf Pine		√
Red Spruce	√	√
American Beech		√
Balsam Fir	√	
Paper Birch	√	
Red Pine	√	
Yellow Birch	√	√
White Spruce	√	
Black Spruce	√	

A second approach is to expand the analysis to focus individually on species with the largest geographical range. Figure 3-11 outlines a much larger geographical area and represents the combined coverage of nine species, which are indicated with a checkmark in the third column of Table 3-10. Each of these species is present in at least nine states. This area also covers a wide range of N+S deposition values (366.1 to 3,100.9 eq/ha/yr). Therefore, this outlined area represents a good region to geographically expand analyses of the impacts of acidifying deposition on biological receptors or endpoints.

The addition of more tree species to the data sets was recommended as the third highest priority because such additions would expand the analyses both comprehensively and geographically. These expanded analyses could potentially provide the basis for an effective tool with which to evaluate the biological and economic impacts of acidifying N+S deposition on biological receptors or end points throughout the United States.

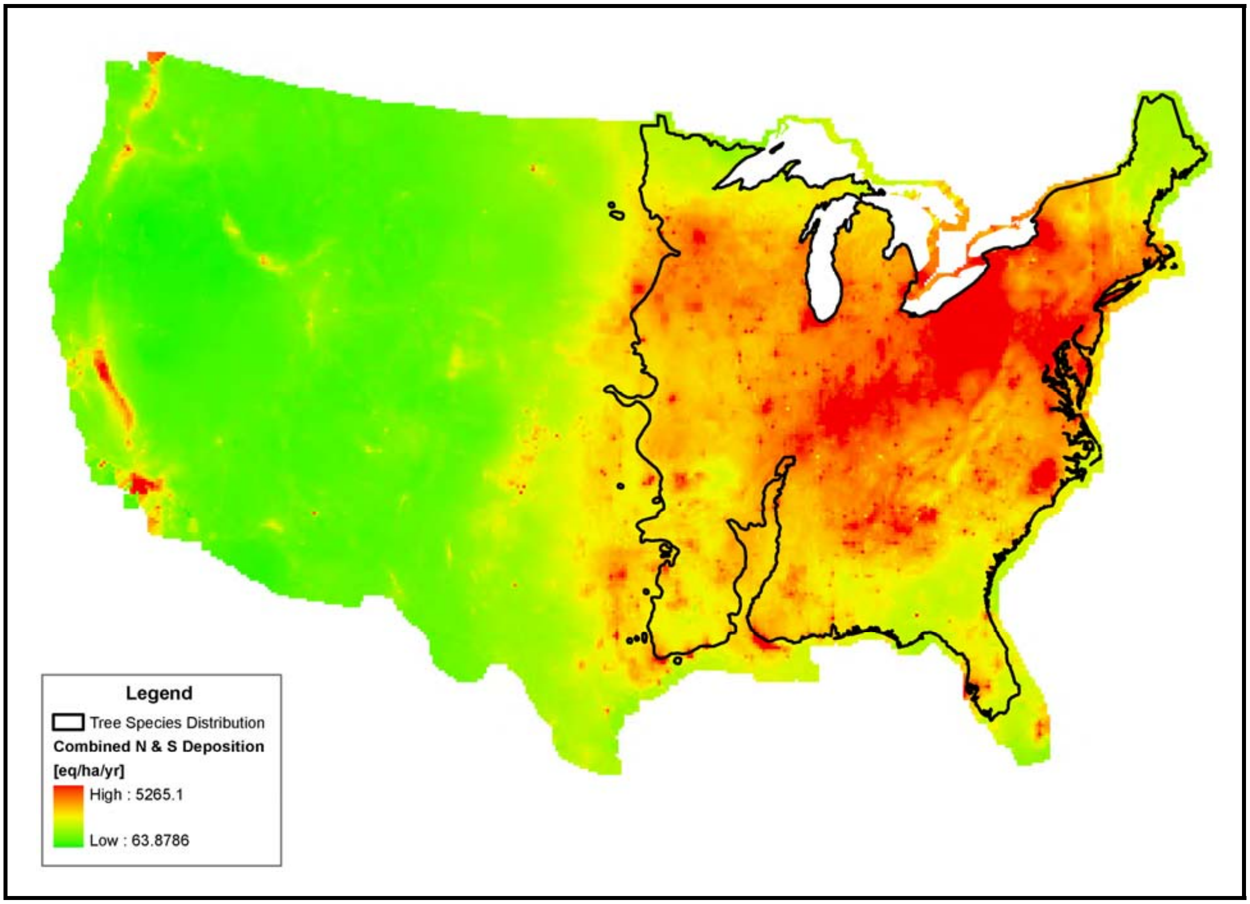

Figure 3-11. The Geographical Area of the Continental United States That Is Occupied by 9 of the 30 Tree Species That Collectively Cover Large Portions of the Midwestern, Southern, and Northeastern Regions of the United States

3.4 References

Alerich, C., USFS, Newtown, PA. Personal communication with Jennifer Phelan, RTI International. May 1, 2009.

Aber, J.D., A. Magill, S.G. McNulty, R.D. Boone, K.J. Nadelhoffer, M. Downs, and R. Hallett. 1995. "Forest Biogeochemistry and Primary Production Altered by Nitrogen Saturation." *Water, Air, and Soil Pollution 85*:1665–1670.

Adams, D., R. Alig, B.A. McCarl, and B.C. Murray. February 2005. FASOMGHG Conceptual Structure, and Specification: Documentation. Available at http://agecon2.tamu.edu/people/faculty/mccarl-bruce/FASOM.html. Accessed on October 22, 2008.

Allen, H.W. 2001. "Silvicultural Treatments to Enhance Productivity." In *The Forests Handbook*, pp. 129–139. Vol. 2. J. Evans, ed. Oxford, UK: Blackwell Science Ltd.

Bechtold, W.A. and P.L. Patterson. 2005. *The Enhanced Forest Inventory and Analysis Program—National Sampling Design and Estimation Procedures*. Gen. Tech. Rep. SRS-80. U.S. Department of Agriculture, Forest Service, Southern Research Station. 85 p.

Brown, L.H. 2002. *Profile of the Annual Fall Foliage Tourist in Vermont: Travel Year 2001.* Report prepared for the Vermont Department of Tourism and Marketing. Burlington, VT: Vermont Tourism Data Center, University of Vermont.

Chapin, F.S., III., L. Moilanen, and K. Kielland. 1993. "Preferential Use of Organic Nitrogen for Growth by a Non-mycorrhizal Arctic Sedge." *Nature 361*:150–153.

Cordell, H.K., C.J. Betz, G. Green, and M. Owens. 2005. *Off-Highway Vehicle Recreation in the United States, Regions and States: A National Report from the National Survey on Recreation and the Environment (NSRE)*. Report prepared for the Forest Service's National OHV Policy & Implementation Teams. USDA Forest Service.

Cordell, K., B. Leeworthy, G.T. Green, C. Betz, and B. Stephens. n.d. *The National Survey on Recreation & the Environment*. Research Work Unit 4953. Athens, GA: Pioneering Research on Changing Forest Values in the South and Nation USDA Forest Service Southern Research Station. Available at www.srs.fs.fed.us/trends.

DeHayes, D.H., P.G. Schaberg, G.J. Hawley, and G.R. Strimbeck. 1999. "Acid Rain Impacts on Calcium Nutrition and Forest Health." *Bioscience 49*(10):789–800.

Driscoll, C.T., K.M., Driscoll, M.J., Mitchell, and D.J. Raynal. 2003. "Effects of Acidic Deposition of Forest and Aquatic Ecosystems in New York State." *Environmental Pollution* 123(3):327-336.

Driscoll, C.T., G.B. Lawrence, A.J. Bulger, T.J. Butler. C.S. Cronan, C. Eager, K.F. Lambert, G.E. Likens, J.L. Stoddard, and K.C. Weathers. 2001. "Acidic Deposition in the Northeastern United States: Sources and Inputs, Ecosystem Effects, and Management Strategies." *Bioscience* 51(3):180–198.

Drohan, J. R., and W.E. Sharpe. 1997. "Long-Term Changes in Forest Soil Acidity in Pennsylvania, U.S.A." *Water, Air, and Soil Pollution* 95:299–311.

Fenn, M.E., T.G. Huntington, S.B. McLaughlin, C. Eager, A. Gomez, and R.B. Cook. 2006. "Status of Soil Acidification in North America." *Journal of Forest Science 52(special issue)*:3–13.

Fox, T., Virginia Polytechnic Institute and State University, Forest Nutrition Cooperative, VA. Personal communication with Jennifer Phelan, RTI International. July 27, 2009.

Haefele, M., R.A. Kramer, and T.P. Holmes. 1991. *Estimating the Total Value of a Forest Quality in High-Elevation Spruce-Fir Forests. The Economic Value of Wilderness: Proceedings of the Conference.* Gen. Tech. Rep. SE-78 (pp. 91-96). Southeastern For. Exper. Station. Asheville, NC: USDA Forest Service.

Holmes, T., and R. Kramer. 1995. "An Independent Sample Test of Yea-Saying and Starting Point Bias in Dichotomous-Choice Contingent Valuation*." Journal of Environmental Economics and Management* 28:121-132.

Holmes, T.P. 1992. *Economic Welfare Impacts of Air Pollution Damage to Forests in the Southern United States*. Asheville, NC: U.S. Dept. of Agriculture, Forest Service, Southeastern Forest Experiment Station. pp. 19-26.

Jenkins, D.H., J. Sullivan, and G.S. Amacher. 2002. "Valuing High Altitude Spruce-Fir Forest Improvements: Importance of Forest Condition and Recreation Activity." *Journal of Forest Economics* 8:77-99.

Johnson, A.H., and T.G. Siccama. 1983. "Acid Deposition and Forest Decline." *Environmental Science and Technology* 17(7): 294A-305A.

Kaval, P., and J. Loomis. 2003. *Updated Outdoor Recreation Use Values With Emphasis On National Park Recreation*. Final Report October 2003, under Cooperative Agreement CA 1200-99-009, Project number IMDE-02-0070.

Killam, K. 1994. *Soil Ecology*. Cambridge, UK: Cambridge University Press.

Kramer, A., T. Holmes, and M. Haefele. 2003. "Contingent Valuation of Forest Ecosystem Protection." In *Forests in a Market Economy*, E.O. Sills and K.L. Abt, eds., pp. 303-320. Dordrecht, The Netherlands: Kluwer Academic Publishers.

Krieger, D.J. 2001. *Economic Value of Forest Ecosystem Services: A Review*. Washington, DC: The Wilderness Society.

Luppold, W.G. 1984. "An Econometric Study of the U. S. Hardwood Lumber Market." *Forest Science* 30(4):1027-1038.

Luzadis, V.A. and E.R. Gossett. 1996. "Sugar Maple." In *Forest Trees of the Northeast*, J.P. Lassoie, V.A. Luzadis, and D.W. Grover, eds., pp. 157-166. Cooperative Extension Bulletin 235. Cornell Media Services. Available at http://maple.dnr.cornell.edu/pubs/trees.htm.

McNulty, S.G., J. Boggs, J.D. Aber, L. Rustad, and A. Magill. 2005. "Red Spruce Ecosystem Level Changes Following 14 Years of Chronic Nitrogen Fertilization." *Forest Ecology and Management* 219:279–291.

McNulty, S.G., E.C. Cohen, H. Li, and J.A. Moore-Myers. 2007. "Estimates of Critical Acid Loads and Exceedences for Forest Soils across the Conterminous United States." *Environmental Pollution* 149:281–292.

Miller, H.G. 1988. "Long-Term Effects of Application of Nitrogen Fertilizers on Forest Sites." In *Forest Site Evaluation and Long-Term Productivity*, pp. 97-106. D.W. Cole and S.P. Gessel, eds. Seattle, WA: University of Washington Press.

Ouimet, R., P.A. Arp, S.A. Watmough, J. Aherne, and I. DeMerchant. 2006. "Determination and Mapping Critical Loads of Acidity and Exceedences for Upland Forest Soils in Eastern Canada." *Water, Air and Soil Pollution* 172:57–66.

Ouimet, R., J.-D. Moore, and L. Duchesne. 2008. "Effects of Experimental Acidification and Alkalinization on Soil and Growth and Health of *Acer saccharum* Marsh." *Journal of Plant Nutrition and Soil Science* 171:868–871.

Pardo, L.H., and N. Duarte. 2007. *Assessment of Effects of Acidic Deposition on Forested Ecosystems in Great Smoky Mountains National Park using Critical Loads for Sulfur and Nitrogen.* U.S. Department of Agriculture, Forest Service. Prepared for Tennessee Valley Authority, S. Burlington, VT.

Perkins, T.D., Adams, G.T., Lawson, S.T., Schabert, P.G., and McNulty, S.G. 2000. "Long-Term Nitrogen Fertilization Increases Winter Injury in Montane Red Spruce (*Picea rubens*)." *Journal of Sustainable Forestry* 10*:* 165-172.

Reed, J.C. and Bush, C.A. 2005. Generalized Geologic Map of the Conterminous United States Edition 1.2. 1:7,500,000 scale. U.S. Geological Survey. Available at http://pubs.usgs.gov/atlas/geologic (accessed on: November 1, 2008).

Rojas, J., Weyerhaeuser Co., MS. Personal communication with Jennifer Phelan, RTI International. May 1, 2006.

Schaberg, P.G., D.H. DeHayes, G.J. Hawley, P.F. Murakami, R.G. Strimbeck, and S.G. McNulty. 2002. "Effects of Chronic N Fertilization on Foliar Membranes, Cold Tolerance, and Carbon Storage in Montane Red Spruce." *Canadian Journal of Forest Research* 32: 1351-1359.

Schaberg, P.G., D.H. DeHayes, G.J. Hawley, G.R. Strimbeck, J.R. Cumming, P.F. Murakami, and C.H. Borer. 2000. "Acidic Mist and Soil Ca and Al Later the Mineral Nutrition and Physiology of Red Spruce." *Tree Physiology* 20: 73-85.

Sheppard, L.J. 1994. "Causal Mechanisms by Which Sulphate, Nitrate and Acidity Influence Frost Hardiness in Red Spruce: Review and Hypothesis." *New Phytologist* 127: 69-82.

Smith, W.B., P.D. Miles, C.H. Perry and S.A. Pugh. 2009. *Forest Resources of the United States, 2007*. Gen. Tech. Rep. WO-78. Washington, DC, U.S. Department of Agriculture, Forest Service, Washington Office.

Spencer, D.M., and D.F. Holecek. 2007. "Basic Characteristics of the Fall Tourism Market." *Tourism Management* 28:491-504.

Sverdrup, H., and P. Warfvinge. 1993a. "Calculating Field Weathering Rates Using a Mechanistic Geochemical Model PROFILE." *Applied Geochemistry* 8:273–283.

Sverdrup, H., and P. Warfvinge. 1993b. *The Effect of Soil Acidification on the Growth of Trees, Grass and Herbs as Expressed by the (Ca+ Mg+ K)/Al Ratio*. Reports in Ecology and Environmental Engineering 2. Lund University, Department of Chemical Engineering, Lund, Sweden.

U.S. Forest Service (USFS). 2006. *Forest Inventory Analysis (FIA) Database*. U.S. Department of Agriculture, Forest Service, Forest Inventory and Analysis National Program, Arlington, VA. Available at http://fia.fs.fed.us/tools-data.

U.S. Forest Service (USFS). 2008. *The Forest Inventory and Analysis Database: Database Description and Users Manual Version 3.0 for Phase 2, revision 1*. Forest Inventory and Analysis Program, U.S. Department of Agriculture Forest Service.

U.S. Department of Agriculture, Forest Service. Forest Inventory and Analysis National Program, RPA Assessment Tables. 2002. Available at http://Ncrs2.Fs.Fed.Us/4801/Fiadb/Rpa_Tabler/Draft_RPA_2002_Forest_Resource_Tables.pdf.

U.S. Department of the Interior, Fish and Wildlife Service, and U.S. Department of Commerce, U.S. Census Bureau. 2007. *2006 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation*.

U.S. Environmental Protection Agency (EPA). 2007. *Assessing the Effects of the Clean Air Act Amendments of 1990 on Ecological Resources: Updated Literature Review and Terrestrial Case Study Approach*. White paper. Washington, DC: U.S. Environmental Protection Agency.

Watmough, S., J. Aherne, P. Arp, I. DeMerchant, and R. Ouimet. 2006. "Canadian Experiences in Development of Critical Loads for Sulphur and Nitrogen." In *Monitoring Science and Technology Symposium: Unifying Knowledge for Sustainability in the Western Hemisphere Proceedings RMRS-P-42CD*, pp. 33–38. C. Aguirre-Bravo, P.J. Pellicane, D.P. Burns, and S. Draggan, eds. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

SECTION 4
AQUATIC ENRICHMENT

4.1 Overview of Affected Ecosystems and Services

One of the main adverse ecological effects resulting from N deposition, particularly in the Mid-Atlantic region of the United States, is the effect associated with nutrient enrichment in estuarine waters. A recent assessment of 141 estuaries nationwide by the National Oceanic and Atmospheric Administration (NOAA) concluded that 19 estuaries (13%) suffered from moderately high or high levels of eutrophication due to excessive inputs of both N and phosphorus, and a majority of these estuaries are located in the coastal area from North Carolina to Massachusetts (NOAA, 2007). By several measures, the aquatic ecosystem of the Chesapeake Bay estuary is particularly suffering from the effects of excessive N loads, and roughly one-third of these loads are associated with atmospheric deposition of N in the watershed (Sweeney, 2007).[1] For other estuaries in the Mid-Atlantic region, the contribution of atmospheric distribution to total N loads is estimated to range between 10% and 58% (Valigura et al., 2001).

Eutrophication occurs in estuaries when nutrient loads exceed the aquatic ecosystems' assimilative capacity for nutrients, and it causes a range of adverse ecological effects. Figure 4-1 provides a conceptual model linking nutrient loadings to specific ecological endpoints and to the main types of ecosystem services affected by eutrophic conditions. When loadings to an estuary exceed its ability to absorb nutrients, they can trigger blooms of phytoplankton and macroalgae, which are primary symptoms of eutrophication in Figure 4-1. These nutrients and the resulting blooms can reduce water clarity and light penetration, which can also result in the loss of submerged aquatic vegetation (SAV). As algal blooms die, they also consume oxygen and lower the amount of available dissolved oxygen (DO) in the water column. Excess nutrients can also lead to more serious "harmful" algal blooms (HABs).

Low DO (i.e., hypoxia) has become a chronic problem in several estuaries, particularly during summer months. Five of the 22 estuaries evaluated by NOAA in the Mid-Atlantic region suffer from serious DO problems. The mainstem of the Chesapeake Bay has been a particular area of concern. For example, between 2005 and 2007, only about 12% of the Bay met DO standards during the summer months (Chesapeake Bay Program, n.d.). Low DO disrupts aquatic habitats, causing stress to fish and shellfish, which, in the short term, can lead to episodic fish

[1] Phosphorus loads, primarily from agricultural runoff and wastewater dischargers in the Chesapeake Bay watershed, are the other main source of nutrients contributing to eutrophication in the Bay.

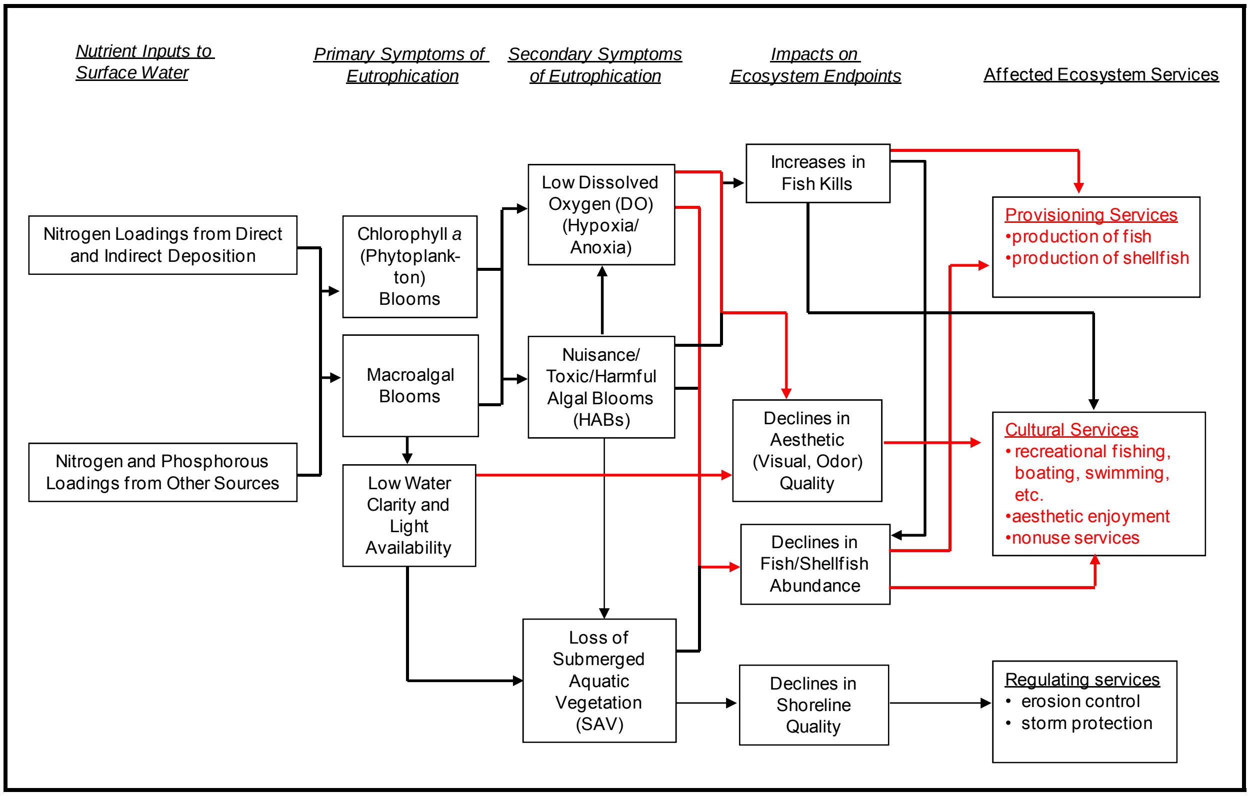

Figure 4-1. Conceptual Diagram of Ecosystem Service Impairments Associated with Aquatic Nutrient Enrichment[a]

Source: Adapted from Bricker et al. (2007) and Bricker, Perreira, and Simas (2003).

[a]Red arrows and fonts highlight the areas for which quantitative models are described in this section. Bold arrows represent the stronger and better established cause-and-effect relationships.

kills and, in the long term, can damage overall growth in fish and shellfish populations. Low DO also degrades the aesthetic qualities of surface water.

HABs were also rated by NOAA as a major problem in five Mid-Atlantic estuaries, including the mainstem of the Chesapeake Bay and the Potomac River estuary. In addition to often being toxic to fish and shellfish and leading to fish kills and aesthetic impairments of estuaries, HABs can, in some instances, also be harmful to human health.

SAV provides critical habitat for many aquatic species in estuaries and, in some instances, can also protect shorelines by reducing wave strength; therefore, declines in SAV due to nutrient enrichment are an important source of concern. Although less prevalent than low DO and HABs as a problematic symptom of eutrophication, it is nonetheless rated by NOAA as a serious problem in the mainstem of the Chesapeake Bay and the New Jersey Inland Bays. It is rated as a moderate problem for the Rappahannock River and Tangier/Pocomoke Sounds, which are part of the Chesapeake Bay watershed, and for Barnegat Bay (New Jersey).

Low water clarity is the result of accumulations of both algae and sediments in estuarine waters. In addition to contributing to declines in SAV, high levels of turbidity also degrade the aesthetic qualities of the estuarine environment. Although NOAA's assessment of estuaries did not separately focus on turbidity as an indicator of eutrophication, it is nonetheless a common problem in the Mid-Atlantic region.

Figure 4-1 also extends the NOAA framework to include links to the main types of ecosystem services that are affected by the primary and secondary symptoms of eutrophication. The following sections provide a discussion and overview of the primarily affected provisioning, cultural, and regulating services.

4.1.1 Effects on Provisioning Services

Estuaries in the eastern United States are an important source of food production, in particular fish and shellfish production. The estuaries are capable of supporting large stocks of resident commercial species, and they serve as the breeding grounds and interim habitat for several migratory species.

To provide an indication of the magnitude of provisioning services associated with coastal fisheries, Table 4-1 reports the annual value of commercial landings in recent years for 15 East Coast states. From 2005 to 2007, the average value of the total catch was $1.5 billion per year. It is not known, however, what percentage of this value is directly attributable to or dependent on the estuaries in these states. Table 4-2 focuses specifically on commercial landings in Maryland and Virginia in 2007, and it reports values for the main commercial species in these states. Although these values also include fish caught outside of the Chesapeake Bay, the values for two key species—blue crab and striped bass—are predominantly from the estuary itself. These data indicate that blue crab landings in 2007 totaled nearly $44 million in the Bay. The value of striped bass and menhaden totaled about $9 million and $25 million, respectively.

To most accurately assess how eutrophication in East Coast estuaries is related to the long-term provisioning services from their fishery resources requires bioeconomic models (i.e., models that combine biological models of fish population dynamics with economic models describing fish harvesting and consumption decisions). In most cases, these models address the dynamic feedback effects between fish stocks and harvesting behavior, and they characterize

Table 4-1. Annual Values of East Coast Commercial Landings (in millions)

State	2004	2005	2006	2007
Connecticut	$33.40	$37.57	$36.89	$42.08
Delaware	$5.42	$6.11	$5.69	$7.58
Florida, East Coast	$39.98	$35.49	$42.00	$42.74
Georgia	$14.37	$13.46	$11.53	$10.08
Maine	$367.09	$391.90	$361.85	$319.52
Maryland	$49.29	$63.67	$53.58	$52.27
Massachusetts	$326.00	$427.07	$437.05	$457.18
New Hampshire	$17.21	$22.12	$18.84	$19.09
New Jersey	$145.86	$159.01	$136.05	$152.46
New York	$46.89	$56.45	$57.73	$58.94
North Carolina	$79.70	$64.89	$70.12	$82.31
Pennsylvania	$0.07	$0.04	$0.10	$0.13
Rhode Island	$76.25	$91.58	$98.58	$76.79
South Carolina	$18.54	$17.57	$17.03	$15.57
Virginia	$160.51	$155.26	$109.07	$130.56
Total	$1,380.60	$1,542.20	$1,456.11	$1,467.31

Source: National Oceanic and Atmospheric Administration (NOAA). (2007, August). "Annual Commercial Landing Statistics." <http://www.st.nmfs.noaa.gov/ st1/commercial/landings/annual_landings.html>.

conditions for a "steady-state" equilibrium, where stocks and harvest levels are stabilized and sustainable over time.

Section 4.2 describes one bioeconomic model linking blue crab harvests to nutrient loads in the Neuse River estuary, and it applies the model to estimate how reductions in N loads to the estuary would affect the societal value of future blue crab harvests. In practice, however, very few other studies have developed empirical bioeconomic models to estimate how changes in environmental quality affect fish harvests and the value of these services (Knowler, 2002). One exception is Kahn and Kemp (1985), which estimated a bioeconomic model of commercial and recreational striped bass fishing using annual data from 1965 to 1979, measuring the effects of SAV levels on fish stocks, harvests, and social welfare. They estimated, for example, that a 50% reduction in SAV from levels existing in the late 1970s (similar to current levels [Chesapeake

Table 4-2. Value of Commercial Landings for Selected Species in 2007 (Chesapeake Bay Region)

State	Species		Value
Maryland			
	Blue crab		$30,433,777
	Striped bass		$5,306,728
	Clams or bivalves		$5,007,952
	Sea scallop		$2,808,984
	Oyster, Eastern		$2,524,045
	Other		$6,190,474
		Total	$52,271,960
Virginia			
	Sea scallop		$62,891,848
	Menhaden		$25,350,740
	Blue crab		$13,222,135
	Croaker, Atlantic		$4,615,924
	Striped bass		$3,834,906
	Clam, Northern Quahog		$3,691,319
	Summer flounder		$3,186,229
	Other		$16,954,893
		Total	$130,561,765

Source: National Oceanic and Atmospheric Administration (NOAA). (2007, August). "Annual Commercial Landing Statistics." <http://www.st.nmfs.noaa.gov/ st1/commercial/landings/annual_landings.html>.

Bay Program, 2008]) would decrease the net social benefits from striped bass by roughly $16 million (in 2007 dollars).

In a separate analysis, Anderson (1989) developed an empirical dynamic simulation model of the effects of SAV changes on commercial blue crab harvests in the Virginia portion of the Chesapeake Bay. Applying the empirical model results, long-run (15-year) dynamic equilibria were estimated under baseline conditions (assuming SAV area constant at 1987 levels) and under conditions with "full restoration" of SAV (i.e., 284% increase). In equilibrium, the increase in annual producer surplus and consumer surplus with full restoration of SAV was estimated to be $3.5 million and $4.4 million (in 2007 dollars), respectively. One study examining the short-term effects of DO levels on crab harvests is Mistiaen, Strand, and Lipton (2003). Focusing on three Chesapeake Bay tributaries—the Patuxent, Chester, and Choptank rivers—they estimated a "stress-availability" model measuring the effects of DO levels on the availability of blue crabs for commercial harvest, given the stock levels and number of fishing vessels. The model results indicated that, below a threshold of 5 mg/L, reductions in DO cause a statistically significant reduction in commercial harvest and revenues. For the Patuxent River alone, a simulated reduction of DO from 5.6 to 4.0 mg/L was estimated to reduce crab harvests by 49% and reduce total annual earnings in the fishery by $275,000 (in 2007 dollars). However, this is an upper-bound estimate because it does not account for changes in fishing effort that would likely occur, and if the measured changes are due to migration of crab populations to other areas rather than to crab mortality, then the broader net effects on crab harvests may also be considerably smaller.[2] Although this study provides useful information regarding the general magnitude of crab fishing losses due to DO, its limitations and the fact that it does not provide an estimate of producer surplus change restrict its suitability for benefits transfer.

In addition to affecting provisioning services through commercial fish harvests, eutrophication in estuaries may also affect these services through its effects on the demand for seafood. For example, a well-publicized toxic pfiesteria bloom in the Maryland Eastern Shore in 1997, which involved thousands of dead and lesioned fish, led to an estimated $56 million (in 2007 dollars) in lost seafood sales for 360 seafood firms in Maryland in the months following the outbreak (Lipton, 1999). Additional evidence regarding potential losses in provisioning services due to eutrophication-related fish kills is provided by Whitehead, Haab, and Parsons (2003) and Parsons et al. (2006). The survey used in both studies was conducted with more than 5,000 respondents in states bordering the Chesapeake Bay area and in North Carolina. The survey asked respondents to consider how their consumption patterns would change in response to news about a large fish kill caused by a toxic pfiesteria bloom. To address the fact that not all fish kills are the same, the size and type of the described fish kill—either "major," involving more than 300,000 dead fish and 75% with pfiesteria lesions, or "minor," involving 10,000 dead fish and 50% with lesions—were randomized across respondents. Based on respondents' stated behaviors, the studies estimated reductions in consumer surplus per seafood meal ranging from $2 to $5.[3] The survey also found that 42% of residents in the four-state area (Maryland, Virginia, Delaware, and North Carolina) were seafood consumers and that the average number of seafood meals per month among these consumers was between four and five. As a result, they estimated

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[2]The estimated relationship between harvest and DO is discontinuous at 5 mg/L. The size of the measured effect on harvests is relatively small below 5 mg/L and zero above the 5 mg/L threshold; therefore, any sizable benefits would require DO to cross the 5 mg/L threshold. Moreover, the 5 mg/L threshold was an assumption of the model rather than a tested hypothesis, which raises additional questions about the accuracy of benefit estimates for changes across the threshold.

[3]Somewhat surprisingly, these estimates were not sensitive to whether the fish kill was described as major or minor or to the different types of information included in the survey.

aggregate consumer surplus losses of $43 million to $84 million (in 2007 dollars) in the month after a fish kill.

4.1.2 *Effects on Cultural Services*

Estuaries in the eastern United States also provide an important and substantial variety of cultural ecosystem services, including water-based recreational and aesthetic services. One of the difficulties with quantifying recreational services from estuaries is that much of the national and regional statistics are jointly collected and reported for estuarine and other coastal areas. Nevertheless, even these combined statistics provide several useful indicators of recreational service flows. For example, data from the FHWAR indicate that, in 2006, 4.8% of the 16 and older population in coastal states from North Carolina to Massachusetts participated in saltwater fishing. The total number of days of saltwater fishing in these states was 26.1 million in 2006. Based on estimates from Kaval and Loomis (2003), the average consumer surplus value for a fishing day was $35.91 (in 2007 dollars) in the Northeast and $87.23 in the Southeast. Therefore, the total recreational consumer surplus value from these saltwater fishing days was approximately $1.28 billion (in 2007 dollars).

Recreational participation estimates for several other coastal recreational activities are also available for 1999 to 2000 from the NSRE. These estimates are summarized in Table 4-3 based on data reported in Leeworthy and Wiley (2001). Almost 22 million individuals aged 16 and older visited beaches in coastal states from North Carolina to Massachusetts, for a total of nearly 209 million days annually during 1999 to 2000. Using a national daily value estimate of $46.91 (in 2007 dollars) for beach visits from Kaval and Loomis (2003), the aggregate value of these coastal outings was $9.8 billion per year. During the same period, almost 7 million people participated in birdwatching, for a total of almost 175 million days (aggregate value of $6.72 billion) per year, and nearly 6 million participated in motorboating for a total of nearly 63 million days (aggregate value of $2.08 billion) per year. More than 3 million participated in visits to nonbeach coastal waterside areas, for a total of more than 35 million days per year. In contrast, fewer than 1 million individuals per year participated in canoeing, kayaking, or waterfowl hunting.

4.1.3 *Effects on Regulating Services*

Estuaries and marshes have the potential to support a wide range of regulating services, including climate, biological, and water regulation; pollution detoxification; erosion prevention; and protection against natural hazards (MEA, 2005). It is more difficult, however, to identify the specific regulating services that are significantly affected by changes in nutrient loadings.

Table 4-3. Participation in Selected Marine Recreation Activities in East Coast States in 1999–2000

State	Swimming		Visiting Beaches		Visiting Watersides Other than Beaches	
	N[a]	Days[b]	N[a]	Days[b]	N[a]	Days[b]
Connecticut	1.06	12.77	1.10	14.07	0.18	2.41
Delaware	0.99	9.77	1.26	12.88	0.08	*
Maryland	2.17	18.35	2.53	18.70	0.47	5.89
Massachusetts	2.74	31.66	2.78	28.68	0.47	2.93
New Jersey	3.80	37.43	3.97	40.88	0.45	4.58
New York	2.39	28.97	2.96	29.23	0.56	3.74
North Carolina	3.22	27.48	3.19	27.94	0.44	4.16
Rhode Island	1.56	19.68	1.43	17.87	0.27	3.31
Virginia	1.70	15.48	2.33	18.75	0.48	8.27
Total:	19.63	201.60	21.54	208.98	3.41	35.29

State	Motorboating		Canoeing	Kayaking	Bird Watching		Waterfowl Hunting
	N[a]	Days[b]	N[a]	N[a]	N[a]	Days[b]	N[a]
Connecticut	0.39	6.76	0.05	0.10	0.45	15.19	0.00
Delaware	0.38	4.56	0.04	0.02	0.43	14.03	0.02
Maryland	0.97	8.13	0.16	0.03	0.82	19.76	0.03
Massachusetts	0.61	6.05	0.07	0.17	1.02	26.10	0.00
New Jersey	0.89	12.45	0.07	0.10	0.80	18.80	0.01
New York	0.90	9.48	0.07	0.06	0.88	24.55	0.00
North Carolina	0.55	7.25	0.04	0.12	1.04	20.52	0.03
Rhode Island	0.38	4.37	0.15	0.11	0.56	19.01	0.00
Virginia	0.60	4.54	0.15	0.06	0.86	17.00	0.04
Total:	5.67	63.59	0.79	0.76	6.84	174.96	0.13

Source: Leeworthy and Wiley (2001)

[a] Number of resident and nonresident participants (in millions).
[b] Number of days by residents and nonresidents (in millions).
[*] insufficient data for estimate

One potentially affected service is provided by SAV, which can help reduce wave energy levels and, thus, protect shorelines against excessive erosion. Declines in SAV may, therefore, also increase the risks of episodic flooding and associated damages to near-shore properties or public infrastructure. In the extreme, these declines may even contribute to shoreline retreat, such that land and structures are lost to the advancing waterline.

4.2 Methodology for Assessing the Benefits of Reductions in Aquatic Enrichments

This section describes methods for valuing changes in several ecosystem services associated with reduced nutrient enrichment effects in estuaries. Primarily because of the availability of existing models and studies, these methods are specifically developed and demonstrated for the Chesapeake Bay and Albemarle Pamlico Sound (including mainly the Neuse River) estuaries. According to the NOAA assessment (NOAA, 2007), both of these estuaries are suffering from high levels of eutrophication, and in both cases, atmospheric deposition accounts for a large portion of total annual nutrient loads to the estuaries.

4.2.1 The Chesapeake Bay Estuary

Based on evidence from several existing valuation studies, this section describes a series of benefit transfer models for estimating the benefits of improved nutrient-related water quality in the Chesapeake Bay. These benefits are associated with enhancements to recreational fishing, boating, and beach use services; aesthetic services; and nonuse services. One important and inherent limitation of these models is that they each require an estimate of water quality change as an input. That is, applying these models first requires a framework for linking changes in atmospheric N deposition in the Chesapeake Bay watershed to changes in water quality conditions in the Bay. Unfortunately, although the case studies in the REA document (EPA, 2009) demonstrate methods for linking changes in deposition to changes in N loadings to the Bay waters, reliable and cost-effective methods for estimating subsequent changes in Bay water quality have not been specified yet. Moreover, each of the benefit transfer models described in this section requires a different measure of water quality as an input. These water quality modeling gaps will need to be addressed before the following benefit transfer models can be fully applied.

4.2.1.1 Recreational Fishing Benefits

This section describes and applies a two-part "benefit transfer" framework for estimating the recreational fishing benefits of improved eutrophic conditions in the Chesapeake Bay.

The first component predicts the effect of changes in average DO levels on recreational fishing catch rates. These catch rates can be interpreted as indicators of the recreational fishing services provided by the Bay. Two catch rate models are described: one based on a study of striped bass fishing in the Bay and the other based on a study of summer flounder fishing in the Maryland coastal bays.

The second component estimates the benefits of catch rate improvements using WTP estimates derived from a meta-analysis study by Johnston et al. (2005) and annual fishing trip estimates to the Bay using data from the Marine Recreation Fishing Statistics Survey (MRFSS).

4.2.1.1.1 Estimating Changes in Recreation Services (Catch Rates) Due to Changes in DO

In two related papers, Lipton and Hicks (1999, 2003) reported the results of a travel cost study of recreational striped bass fishing in the Chesapeake Bay. One of the main focuses of the study was measuring the effect of DO levels on striped bass catch rates. The fishing data for this study were drawn from the National Marine Fisheries Service's (NMFS's) 1994 MRFSS, which included 407 intercept sites in the Bay and 1,806 striped bass angler respondents. The DO water quality data were from biweekly summer sampling at 207 locations in the Bay.

The striped bass catch model assumes that the number of fish caught per trip (in logarithmic form) at a site is a linear function of several factors, including the hours spent by the angler at the site on the trip, the angler's experience and skill in saltwater fishing, and water quality conditions at the site. Water quality is characterized in the model by surface temperature (*ST*), bottom temperature (*BT*), surface DO (*SDO*), and bottom DO (*BDO*). According to the functional form of the estimated model, the *change* in the expected striped bass catch rate per trip due to a water quality change can be expressed as

$$\Delta Q = Q_1 - Q_0 = \exp(f_B(\Delta WQ) + \ln Q_0) - Q_0, \tag{4.1}$$

where Q_i is the expected number of striped bass caught per trip under conditions i, such that $i = 0$ represents reference conditions and $i = 1$ represents conditions after the water quality change. The function $f_B(\Delta WQ)$ represents the combined effect of changes in temperature and DO on expected catch rates. Using the parameter estimates from the empirical catch rate model, this function for striped bass can be expressed as

$$\begin{aligned} f_B(\Delta WQ) = \ln Q_1 - \ln Q_0 &= -.0.2548(ST_1 - ST_0) + 0.3225(BT_1 - BT_0) \\ &+ 0.2589(SDO_1 - SDO_0) + 0.2253(BDO_1 - BDO_0) - 0.0167(BDO_1^2 - BDO_0^2) \end{aligned} \tag{4.2}$$

To demonstrate the model, we quantified reference-level catch rates (Q_0) using recent MRFSS data for the Bay, which are summarized in Table 4-4. The table reports average catch

Table 4-4. Average Catch Rate per Fishing Trip in the Chesapeake Bay, by State and Targeted Fish Species

Fishing Trip	2001	2002	2003	2004	2005	Average 2001–2005
Maryland residents						
Striped bass	1.20	1.58	1.99	1.81	1.70	1.65
Summer flounder	0.09	0.08	0.09	0.34	0.04	0.12
Other species	0.29	0.32	0.32	0.27	0.45	0.33
All species	0.34	0.38	0.40	0.35	0.51	0.39
Virginia residents						
Striped bass	0.42	0.44	0.52	0.82	0.68	0.59
Summer flounder	0.96	0.80	0.91	0.93	0.69	0.86
Other species	0.34	0.40	0.26	0.27	0.34	0.32
All species	0.37	0.42	0.29	0.32	0.36	0.35

Source: National Oceanic and Atmospheric Administration (NOAA), 2009.

rates for striped bass and other key recreational species for 2001 to 2005. Over the 5-year period, striped bass catch rates averaged 1.65 fish per trip in Maryland and 0.59 fish per trip in Virginia.[6]

With these reference condition catch rate estimates, Equations (4.1) and (4.2) can be used to predict the change in average catch rate (ΔQ) associated with specific changes in surface and bottom temperature and DO levels (i.e., those associated with regulatory baseline and regulatory control scenarios). For example, if average surface and bottom DO levels in the Bay both increase by 2.41 units (with no change in temperature), the striped bass catch rate is predicted to increase by 1.57 in Maryland and by 0.56 in Virginia (a 94.9% increase).

It is more difficult to develop catch rate predictions for other recreational species, because of the apparent lack of any other empirical studies that have estimated the relationship between water quality conditions and recreational catch rates in the Bay.[7] One alternative is to assume that the striped bass model described above is applicable to other species; however, the resulting catch rate change estimates would inevitably have higher levels of uncertainty associated with them.

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[6]For comparison, Lipton and Hicks (1999) reported that average catch rates in 1994 were 0.71 in Maryland and 0.66 in Virginia.

[7]Bricker et al. (2006) described similar models for the Potomac and Patuxent River estuaries and other East Coast estuaries; however, they did not provide parameter estimates for these models.

A second approach is to use catch rate models developed in areas outside the Bay; however, only one such study was found.[8] Massey, Newbold, and Gentner (2006) used data from the Maryland coastal bays to estimate a catch rate model for recreational summer flounder fishing. They found significant effects from *DO*, temperature (*T*), and water clarity (secchi depth [*SD*]) on recreational catch. Using the parameter estimates from this model, the following function summarizes the measured effects of water quality on summer flounder catch rates:

$$f_F(\Delta WQ) = 0.117(DO_1 - DO_0) + 0.126(T_1 - T_0) + 1.392(SD_1 - SD_0)\,. \tag{4.3}$$

Applying this function to Equation (4.1) in place of $f_B(\Delta WQ)$, a 2.41-unit increase in *DO* (with no change in *T* or *SD*) is predicted to increase summer flounder catch by an additional 0.04 fish per trip in Maryland and 0.28 fish per trip in Virginia (a 32.6% increase). Transferring this model from the Maryland coastal bays to the Chesapeake Bay also contributes to the uncertainty in catch rate predictions for summer flounder, although arguably less so than transferring models from other species (i.e., striped bass) within the Bay.

4.2.1.1.2 Valuing Changes in Catch Rates

The second component of the proposed benefit transfer model for recreational fishing can be summarized as follows:

$$AggB_{fisht} = \Sigma_j\,(WTP_{fish} \times T_{jt}) \times \Delta Q_{jt}, \tag{4.4}$$

where

$AggB_{fisht}$ = aggregate annual benefits in year *t* (in 2007 dollars) to Chesapeake Bay anglers for specified increases in species-specific average catch rates per trip (ΔQ_j, where *j* is the species indicator)

ΔQ_{jt} = predicted change in average catch rate (number of fish caught) per trip for species *j* in the Chesapeake Bay (as described in Section 4.2.1.1.1) in year t

WTP_{fish} = average WTP per additional fish caught per trip

T_{jt} = total number of annual fishing trips (in year *t*) targeting species *j* in the Chesapeake Bay

[8] Kaoru, Smith, and Liu (1995) also estimated the effects of estuarine water quality on recreational fishing in North Carolina; however, rather than using ambient water quality measures, they used estimates of nutrient and biochemical oxygen demand loadings as proxies for water quality conditions.

A large number of revealed- and stated-preference studies have estimated welfare changes associated with changes in recreational fishing catch rates in the United States. Most of these results have been synthesized in a meta-analysis study by Johnston et al. (2006), which estimated meta-regression models controlling for differences across studies in type of water resource, context, angler attributes, and in-study methods. Using these summary models, they predicted average WTP per fish per trip for different species categories. For both Atlantic small game (including striped bass) and Atlantic flatfish (including summer flounder), they predicted WTP ranging from $3 to $11 in 2003 dollars. This meta-analysis study included one WTP estimate from a Chesapeake Bay striped bass study (Bockstael, McConnell, and Strand, 1989), which falls slightly below this range ($2.23), but it did not include a more recent striped bass estimate from the Lipton and Hicks (1999) study, which falls within the upper end of the range ($10.91). Johnston et al.'s (2006) study also did not include the estimate for summer flounder in Maryland coastal bays from Massey, Newbold, and Gentner (2006), which falls within the lower end of the range ($4.22 in 2002 dollars).

Based on these WTP results from the literature, a value range of $2.50 to $12.50 for WTP_{fish}, with a midpoint of $7.50, was selected.

To quantify annual trips by species (T_j), recent MRFSS data for the Bay, which are summarized in Table 4-5, can be used again. The table reports total annual trips for striped bass and other key recreational species from 2001 to 2005. To approximate trips in 2020 (and beyond), one approach is to calculate the average number of trips from 2001 to 2005 by species (as shown in Table 4-5) and increase these values by the expected state-level population growth from 2003 to 2020 (and beyond).

4.2.1.1.3 Limitations and Uncertainties

Although the objective of the previously described approach is to make the best use of existing research to quantify the relationship between changes in eutrophic conditions and recreational fishing benefits in the Bay, the following limitations and uncertainties must also be noted.

First, the catch rate models summarized in Equations (4.2) and (4.3) are most likely to understate the effects of long-term changes (i.e., over several years) in water quality across the entire Bay. Both models are based on analyses that use spatial and short-term (during a single year's fishing season) temporal variation to measure the relationship between catch rates and water quality conditions. Therefore, these measured relationships cannot be expected to capture

Table 4-5. Aggregate Number of Fishing Trips to the Chesapeake Bay, by State and Targeted Fish Species

Fishing Trip	2001	2002	2003	2004	2005	Average 2001–2005
Maryland residents						
Striped bass	2,594,971	2,014,818	2,579,771	2,176,824	2,351,145	2,343,506
Summer flounder	2,106,810	1,268,048	1,598,484	1,486,154	1,734,101	1,638,719
Other species	33,457,937	31,349,971	48,352,248	39,740,106	34,503,965	37,480,845
All species	38,159,717	34,632,837	52,530,503	43,403,083	38,589,211	41,463,070
Virginia residents						
Striped bass	2,043,025	1,911,180	2,369,576	2,525,057	2,549,248	2,279,617
Summer flounder	2,285,628	1,982,130	2,300,633	2,556,902	2,549,248	2,334,908
Other species	49,915,214	47,535,158	67,839,883	65,345,054	58,036,434	57,734,349
All species	54,243,868	51,428,468	72,510,092	70,427,013	63,134,930	62,348,874

Source: National Oceanic and Atmospheric Administration (NOAA), 2009.

the dynamic effects of long-term changes in DO on the overall growth and abundance of the striped bass and summer flounder populations in the Bay.

Second, as previously noted, empirical catch rate models are only available for striped bass and summer flounder, and the model for the latter species is based on data from outside the Bay. Although it is not difficult to apply these models to estimate catch rate changes for other species within the Bay, the resulting estimates are subject to significant uncertainty, because there is little evidence about how well these models transfer to other species.

Third, the valuation model summarized in Equation (4.4) uses a number of simplifying assumptions. In particular, the value per fish caught is assumed to be constant, but within a large range—\$2.50 to \$12.50—that can significantly affect the aggregate benefit estimates. In addition, the total number of fishing trips is assumed to be unaffected by changes in catch rates. This restriction is expected to understate the true aggregate benefits of increased catch rates, because higher catch rates would most likely increase the number of fishing trips.

4.2.1.2 Recreational Boating Benefits

To estimate benefits to Chesapeake Bay boaters, a benefit transfer approach that uses value estimates developed by Lipton (2004) is described. That study used a CV method and survey data from 755 Maryland boaters in 2000 to estimate the individual and aggregate benefits of a 1-unit improvement in respondents' water quality rating (on a 1 to 5 scale from "poor" to "excellent") for the Bay. The benefit transfer model based on this study can be summarized as follows:

$$AggB_{boatt} = \Sigma_i \, \Sigma_j(WTP_{boat,i} \times N_{i,j} \times b_j \times g_{tj}) \times \Delta WQ_5, \tag{4.5}$$

where

ΔWQ_{5t} = change in Chesapeake Bay water quality, expressed on a 5-point rating scale (from "poor" to "excellent") in year t

$AggB_{boatt}$ = aggregate annual benefits in year t (in 2007 dollars) to Maryland, Virginia, and DC boat owners who use the Chesapeake Bay as their principal boating area for a specified ΔWQ_5 increase in water quality

$WTP_{boat,i}$ = average annual WTP (in 2007 dollars) per boater for a 1-unit increase in water quality on the WQ_5 scale (i = sailboat, trailered powerboat, or in-water powerboat)

$N_{i,j}$ = total number of boats by type i and location j (j = Maryland, Virginia, or DC) of boat ownership in 2007

b_j = the ratio of (1) registered boat *owners* whose principal boating area is the Chesapeake Bay to (2) the total number of registered *boats* (by location j)

g_{tj} = population growth factor for location j from 2007 to year t

Lipton (2004) reported estimates of average WTP by boat owners in three different categories for a 1-unit increase in water quality ($\Delta WQ_5 = 1$) in the Chesapeake Bay. Sailboat owners had the highest average WTP of $93.26 (in 2000 dollars). Trailered and in-water powerboat owners had an average WTP of $30.25 and $77.98, respectively.

Converting these Lipton (2004) estimates to 2007 dollars with the consumer price index (CPI) results in WTP_{boat} estimates of $112.29, $36.42, and $93.89 for sailboat, trailered powerboat, and in-water powerboat owners, respectively (Table 4-6).

N_{MD} was estimated for the three boater categories using data on Maryland boat ownership from Lipton (2006, 2008). Lipton (2008) quantifies sailboat and powerboat ownership for 2007, but it does not break out powerboats according to whether they were trailered or in-water boats. To develop separate estimates for these two subcategories, the proportions reported for 2005 (Lipton, 2006), which indicated that 79.8% of powerboats in Maryland were trailered, were applied. To estimate N_{VA} and N_{DC}, the total number of registered boats in Virginia and DC in 2006 was obtained from the National Marine Manufacturers Association (2008), and this number

Table 4-6. Input Estimates for the Chesapeake Bay Boating Benefit Transfer Model

	Number of Registered Boats			Adjustment Factor			
Boat Type	N_{MD}	N_{VA}	N_{DC}	b_{MD}	b_{VA}	b_{DC}	WTP_{boat}
Sailboat	8,200	9,200	100	60.76%	56.92%	60.76%	$112.29
Trailered powerboat	93,300	104,600	1,100	60.77%	56.93%	60.77%	$36.42
In-water powerboat	23,600	26,400	300	60.77%	56.93%	60.77%	$93.89
Total	125,100	140,300	1,500				

was augmented by the observed growth rate in Maryland boat ownership from 2006 to 2007. To separate these total numbers into the three categories of boat ownership, the same proportions estimated for Maryland registered boats in each category were applied.

The value b_{MD} represents a two-part adjustment to the total number of registered boats in Maryland, as estimated by Lipton (2004). The first converts the total number of registered boats to the total number of boat owners, because some boat owners own more than one boat. The second adjusts for the fact that, for some Maryland boaters, the Chesapeake Bay is not their principal boating area. Every 100 registered boats correspond to an estimated 60.8 boat owners whose principal boating area is the Chesapeake Bay. The same adjustment factor for registered boaters in DC was applied to estimate b_{DC}.

To estimate b_{VA}, the expected 6.3% of registered boats in Virginia Beach (Murray and Lucy, 1981), which is the main Virginia coastal area outside the Bay, was first excluded; then the same adjustment factor developed for Maryland and DC was applied. Thus, in Virginia, for every 100 registered boats, there are 56.9 boat owners whose principal boating area is the Chesapeake Bay.

To estimate benefits in future years, it is assumed that the number of boaters per state increases at the same rate as population growth. This growth rate from 2007 to 2020 (and beyond) is included in the factor g_{jt}.

4.2.1.2.1 Limitations and Uncertainties

A potential limitation of the proposed benefit transfer model for boating services is the uncertainty associated with directly translating estimated water quality changes due to changes in nutrient loadings into the WQ_5, which is a subjective index based on boaters' perceptions and experience. These perceptions may be based, at least in part, on observations unrelated to eutrophic conditions (e.g., trash in the water or advisories based on pathogen levels).

The other main source of uncertainty is with the number of affected boaters. As in the recreational fishing model, the affected number of recreators is assumed to be unaffected by the change in water quality. This assumption is likely to lead to an underestimate of the aggregate benefit to boaters of a water quality improvement.

One alternative approach is to use value estimates from Bockstael, McConnell, and Strand (1989), who also estimated changes in consumer surplus for trailered boat owners in Maryland resulting from a 20% decrease in the product of total N and phosphorus (TNP) levels in the Bay. By rescaling and updating their estimates to 2007 dollars, the implied average WTP per Maryland trailered boat owner per 1% decrease in TNP is $5.38. Applying this value to the estimated total number of trailered powerboat owners in Maryland, Virginia, and DC (see Table 4-6) implies that the aggregate benefits to these boaters per 1% decrease in TNP in the Bay would be $120,000. The main advantage of this approach compared with the model summarized in Equation (4.5) is that it is based on an objective measure of water quality. The fact that it is based on values estimated through a revealed-preference travel cost model of actual boating behavior, compared with a stated-preference CV approach, may be seen as an advantage. However, this approach also has several drawbacks: (1) it is based on considerably older data (from 1984), (2) it only includes direct estimates for trailered boaters, and (3) it includes a potentially narrower measure of value than the Lipton (2004) study because it uses revealed- rather than stated-preference data.

4.2.1.3 Beach Use Benefits

To estimate benefits to Chesapeake Bay beach users, the benefit transfer approaches developed by Morgan and Owens (2001) and Krupnick (1988) were adapted and updated. Both of these studies estimated the aggregate benefits to Maryland, Virginia, and DC households of percentage reductions in levels in the Bay. The fundamental benefit transfer model can be summarized as follows:

$$AggB_{beacht} = (WTP_{beach} * (N_1 b_1 g_{1t} + N_2 b_2 g_{2t}) * t_{beach}) * \Delta\%WQ_{TNPt}, \tag{4.6}$$

where

$\Delta\%WQ_{TNPt}$ = percentage change in Chesapeake Bay water quality in year t, expressed in terms of the average TNP levels, each measured in parts per million (ppm)

$AggB_{beacht}$ = aggregate annual benefits in year t (in 2007 dollars) to Maryland, Virginia, and DC households for a specified $\Delta\%WQ_{TNP}$ increase in water quality in the Bay

WTP_{beach} = average annual household WTP (in 2007 dollars) per trip for a 1% reduction in TNP levels in the Bay

N_1 = total number of households in the 1980 Baltimore and DC standard metropolitan statistical areas (SMSA) in 2007

N_2 = total number of Maryland and Virginia households outside the SMSA in 2007

b_1 = portion of SMSA households with at least one Chesapeake Bay beach trip in the year

b_2 = portion of non-SMSA households in Maryland and Virginia with at least one Chesapeake Bay beach trip in the year

g_1 = population growth factor from 2007 to year t for SMSA household;

g_2 = population growth factor from 2007 to year t for non-SMSA households in Maryland and Virginia

t_{beach} = average number of Chesapeake Bay beach trips per year for beach-going Maryland, Virginia, and DC households

Table 4-7 summarizes value estimates for these model components. Values for WTP_{beach} were derived using estimates from Bockstael, McConnell, and Strand (1988, 1989). Using data from 408 summer beach users in 1984 at nine Maryland western shore beaches and average county-level summer TNP values, they estimated a varying parameter travel cost model. Based on the model results, they reported *aggregate* annual consumer surplus gains of \$34.66 million (in 1987 dollars) for beachgoers residing in the SMSA associated with a 20% decrease in TNP in the Bay. The study also reported that (1) 401,000 SMSA households per year (in the early 1980s) visited Chesapeake Bay beaches and (2) the average number of trips per year for these beach-going households was 4.35,[9] which implies that there were an estimated 1,745,000 trips to the Bay by SMSA households in 1984. Dividing the aggregate benefit estimate by this number of trips implies an average per-trip benefit of \$19.86 (in 1987 dollars) for a 20% reduction in TNP.

[9]This number is actually inferred from a description of values Bockstael, McConnell, and Strand (1989) derived from an alternate model. The value per household user (\$4.70) was divided by the value per trip (\$1.08) to get trips per household (4.35).

Table 4-7. Input Estimates for the Chesapeake Bay Beach-Use Benefit Transfer Model

Beach Use	Number of Households (*N*)	Percentage of Bay Beachgoers (*b*)	Average Beach Trips per Year (*t*)	WTP_{beach}
SMSA	2,744,217	21.00%	4.35	$1.81
Non-SMSA	2,540,214	3.08%	4.35	$1.81
Total	5,284,431	12.38%	4.35	$1.81

To estimate WTP_{beach}, the $19.86 estimate was divided by 20 (i.e., it was assumed that each percentage reduction in TNP has the same value), and the estimate was converted to 2007 dollars using the CPI to adjust for inflation. The resulting estimate for WTP_{beach} is $1.81.

N_1 and N_2 were estimated using the Census estimates of population by county in 2007, multiplied by the ratio of households to population by county in the 2000 U.S. Census. From this calculation, it was estimated that a total of 5.28 million households are in Maryland, Virginia, and DC, and 2.74 million of these are within the SMSA.

For b_1, the Bockstael, McConnell, and Strand (1989) estimate that 21% of households in the SMSA take at least one beach trip to the Chesapeake Bay a year was applied. To derive b_2, this estimate was combined with data from the 2006 Virginia Outdoors Survey (Virginia Department of Conservation and Recreation, 2007), which reports that 8% of *all* the households in Virginia take at least one beach trip to the Chesapeake Bay (or other tidal bays) per year. Taken together, these estimates imply that approximately 3% of *non-SMSA* Virginia households take at least one beach trip per year to the Bay. Applying this estimate to Maryland non-SMSA households as well, it was assumed that b_2 equals 3%.

To estimate t_{beach}, the Bockstael, McConnell, and Strand (1989) estimate of 4.35 trips per year was applied, recognizing that it is most likely an overestimate for non-SMSA beach-going households.

4.2.1.3.1 Limitations and Uncertainties

One of the main limitations of the beach-use valuation model described above is that it is based on value estimates that are from 1984 and, therefore, may be outdated. Beach conditions and recreator preferences in the Bay may have changed significantly since then. In addition, several uncertainties are associated with the estimated number of beach trips by Maryland, Virginia, and DC households in 2007. These estimates are based on limited and, in some cases, relatively old data regarding the percentage of households in each state that use the Bay's beaches and the average number of annual beach trips for those who do.

4.2.1.4 Aesthetic Benefits

To estimate the benefits of improved aesthetic services due to improvements in Chesapeake Bay water quality, a benefit transfer model that is based on estimates of near-shore residents' values for small water-quality changes was developed and applied. The transfer function has the following form:

$$AggB_{homet} = \Sigma_k\, MWTP_k \times \Delta DIN_{kt} \times N_k, \times g_{kt} \tag{4.7}$$

where

- ΔDIN_{kt} = reduction in dissolved inorganic nitrogen (DIN) levels in year t in the portion of the Chesapeake Bay closest to coastal Census block group k
- $AggB_{homet}$ = aggregate annual benefits in year t (in 2007 dollars) to homeowners in all Chesapeake Bay coastal block groups for specified ΔDIN_k changes in water quality
- N_k = estimated number of specified owner-occupied homes in block group k in 2007
- g_{kt} = population growth factor for block group k from 2007 to year t
- $MWTP_k$ = estimated annual marginal WTP (in 2007 dollars) for a 1-unit reduction in water quality, $\Delta DIN_k = 1$, in block group k

To parameterize this function, results from a hedonic housing price study by Poor, Pessagno, and Paul (2007) were used. Using data on 1,377 residential home sales from 1993 to 2003 in St. Mary's River watershed in Maryland, this study regressed the natural log of real home prices (in 2003 dollars) against structural, neighborhood, and environmental water quality characteristics. It specifically estimated the effect of differences in DIN (mg/L), as measured by the annual average in the year of sale at the closest water monitoring station, on log home prices.[10] The study found a statistically significant effect with a model coefficient estimate of −0.0878.

[10] A separate model reported in Poor, Pessagno, and Paul (2007) used total suspended solids (mg/L) instead of DIN as the water quality measure. It was also found to have a statistically significant effect on home prices.

To convert this semielasticity coefficient, which measures the marginal effect of DIN on the log of home price, to $MWTP_k$, which represents the *annualized* average *dollar* value of a 1-unit reduction in DIN for homes in block group k, the following conversion equation was used:

$$MWTP_k = 0.0878 * P_k * A(i,T)\,, \tag{4.8}$$

where

P_k = average price of specified owner-occupied homes in block group k

A = annualization factor, which is a function of the assumed interest rate (r) and average lifetime of homes in years (T).[11] For $r = 0.05$ and $T = 50$, $A = 0.0522$.

To implement the model, Chesapeake Bay coastal block groups were defined as those block groups with a Chesapeake Bay coastline, as delineated by the Census block group boundary files (Environmental Systems Research Institute, Inc. [ESRI], 2002), as well as those block groups whose geographic centroids are located within 1 mile of the coast. This second condition was added to ensure that a majority of the included properties are located within roughly 2 miles of the coast. As shown in Figure 4-2, 1,066 block groups met these criteria.

Within these block groups, the study focused on Census "specified owner-occupied housing units," which include only single-family houses on fewer than 10 acres without a business or medical office on the property. These properties match best with the types of properties analyzed in the hedonic study described above, and the decennial Census provides both count and property value estimates for these homes. Thirty-six of the identified 1,066 block groups had no specified owner-occupied homes and were excluded from the analysis.

To estimate N_k, the number of specified owner-occupied homes in each block group in 2000 was augmented by the growth rate in housing units in the block group's county from 2000 to 2007 (U.S. Census Bureau, 2008b). To estimate the number of specified owner-occupied housing units per block group in future years, an additional population growth factor (g_{kt}) must be applied, for example, by applying county-level population growth rate projections.

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[11] $A(r,T) = 1 \Big/ \left(\sum_{t=1}^{T} (1+r)^{-(t-1)} \right)$

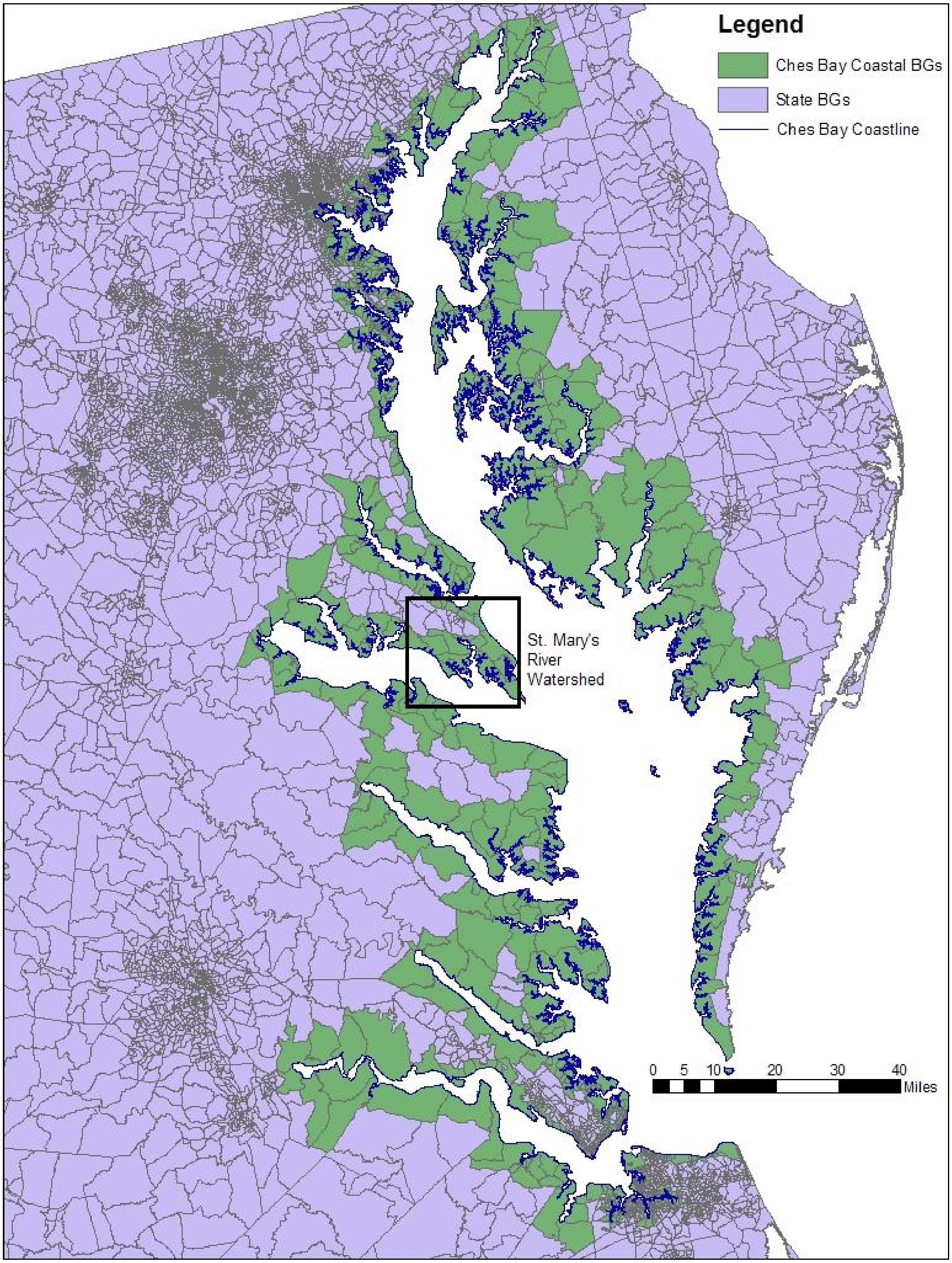

Figure 4-2. Chesapeake Bay Coastal Block Groups

To estimate P_k, the average price of specified owner-occupied homes in 2000 in each block group was adjusted to 2007 using the CPI-Shelter values for Washington-Baltimore, DC-MD-VA-WV.[12] Table 4-8 summarizes the estimated values for N_k and P_k.

4.2.1.4.1 Limitations and Uncertainties

Many of the limitations and uncertainties surrounding this benefit transfer model are associated with the limitations and uncertainties inherent in the hedonic "implicit price" estimate, $MWTP_k$. From a strictly conceptual standpoint, the hedonic implicit price provides a correct measure of the welfare gains to residents of relatively *small* and *localized* improvements in the amenity, in this case changes in DIN water quality. However, caution is required when using this implicit price to estimate the benefits of either a large water quality change or a change that affects many housing consumers. The accuracy of the benefit transfer model summarized by Equation (4.7) will tend to decline as the value of ΔDIN_k increases and as N_k increases. This is because changes that are larger and that affect more consumers are also more likely to cause shifts in the housing market, resulting in potentially large transaction (e.g., moving) costs and changes in the market price equilibrium. Nevertheless, Bartik (1988) has shown that, under many common conditions, models such as Equation (4.7) can be interpreted as providing an upper-bound estimate of aggregate benefits.

From an empirical standpoint, there are other potential limitations and uncertainties. First, there are potential errors in the hedonic parameter estimate. For example, *DIN* may be correlated with other influential housing or neighborhood characteristics that are not included in the hedonic model, in which case the parameter estimate is likely to overstate the implicit price of DIN. Second, for this benefit transfer model, it was assumed that the Census block groups along the Chesapeake Bay coast represent the areas in which the hedonic estimates can most reasonably be applied; however, this spatial extrapolation has inherent limitations. In particular, the implicit price estimates are expected to be less accurate as a measure of WTP in areas that are farther from the hedonic study area (e.g., St. Mary's River watershed), particularly areas that are more urban and densely populated. By excluding homes in other noncoastal Census block groups that are also near the Bay, the benefit transfer model is also likely to exclude some beneficiaries of improved aesthetic services and, therefore, underestimate aggregate benefits. Third, the implicit price was measured using data on individual homes and water quality measures within at

[12] In the decennial Census, values of specified owner-occupied homes were grouped into ranges of values (e.g., from $250,000 to $300,000). With the exception of the highest range, which is $1,000,000 and greater (no upper bound), the midpoint of each range was used to calculate the mean value for each block group. For the highest range, a central value of $1,250,000 was selected.

Table 4-8. Summary of Housing Unit Numbers and Average Prices in Chesapeake Coastal Block Groups in 2007

State, County	Number of Coastal Block Groups	Number of Specified Single-Unit Dwellings per Block Group (N_k)				Average Value of Specified Units per Block Group (P_k)			
		Mean	Std. Dev.	Min	Max	Mean	Std. Dev.	Min	Max
Maryland									
Anne Arundel County	163	470	258	13	1,604	$298,348	$128,985	$81,409	$686,879
Baltimore County	101	339	160	16	859	$154,975	$54,654	$65,209	$343,455
Calvert County	25	583	380	190	1,957	$257,582	$68,476	$174,459	$498,410
Caroline County	2	346	37	310	383	$145,589	$3,499	$142,091	$149,088
Cecil County	18	408	206	195	1,088	$206,114	$41,341	$138,614	$270,798
Charles County	6	466	208	302	909	$263,744	$46,708	$196,436	$319,093
Dorchester County	20	263	125	46	584	$167,317	$61,534	$86,191	$310,723
Harford County	23	397	296	20	926	$187,685	$39,578	$116,187	$303,797
Kent County	13	294	105	55	418	$225,531	$48,381	$143,163	$306,886
Prince George’s County	1	175	0	175	175	$296,739	$0	$296,739	$296,739
Queen Anne’s County	18	584	249	158	1,104	$277,352	$73,004	$153,496	$478,396
St. Mary’s County	29	458	195	92	910	$252,720	$41,564	$174,343	$317,879
Somerset County	14	258	118	81	528	$130,391	$30,635	$77,591	$181,198
Talbot County	20	431	181	192	943	$341,180	$162,586	$128,864	$658,874
Wicomico County	7	304	129	163	576	$145,763	$29,367	$110,355	$186,045
Worcester County	1	148	0	148	148	$98,596	$0	$98,596	$98,596
Baltimore city	116	168	95	10	436	$106,483	$52,800	$23,628	$345,380
Virginia									
Accomack County	9	247	93	128	388	$127,159	$56,731	$69,539	$236,974
Charles City County	4	270	46	219	340	$163,344	$43,411	$104,671	$227,234
Essex County	5	255	69	135	348	$175,598	$52,691	$132,168	$278,618
Gloucester County	16	438	248	174	1,063	$194,628	$40,091	$127,058	$285,437
Isle of Wight County	10	515	204	292	1,076	$210,389	$42,265	$136,806	$273,345

(continued)

Table 4-8. Summary of Housing Unit Numbers and Average Prices in Chesapeake Coastal Block Groups in 2007 (continued)

State, County	Number of Coastal Block Groups	Number of Specified Single-Unit Dwellings per Block Group (N_k)				Average Value of Specified Units per Block Group (P_k)			
		Mean	Std. Dev.	Min	Max	Mean	Std. Dev.	Min	Max
James City County	7	817	551	85	1,734	$306,926	$173,652	$88,149	$569,755
King and Queen County	2	200	7	193	206	$121,113	$3,296	$117,817	$124,409
King George County	3	227	175	3	431	$193,030	$31,483	$151,588	$227,847
King William County	1	398	0	398	398	$128,864	$0	$128,864	$128,864
Lancaster County	11	325	78	145	447	$255,256	$51,809	$157,850	$339,499
Mathews County	4	700	335	279	1,131	$203,309	$28,957	$153,167	$220,637
Middlesex County	9	301	88	138	415	$205,784	$36,392	$146,531	$249,250
New Kent County	1	497	0	497	497	$188,507	$0	$188,507	$188,507
Northampton County	9	267	48	178	365	$151,509	$34,176	$100,355	$214,862
Northumberland County	9	363	124	219	670	$233,922	$35,330	$182,242	$303,884
Prince George County	4	362	220	88	591	$197,887	$46,549	$144,641	$269,254
Richmond County	3	307	33	276	353	$172,231	$19,788	$157,244	$200,192
Surry County	2	317	67	250	384	$172,904	$12,214	$160,690	$185,119
Westmoreland County	13	311	83	181	458	$161,391	$32,705	$98,923	$228,513
York County	14	549	257	97	1,114	$239,291	$54,067	$123,412	$333,513
Chesapeake city	29	347	200	38	747	$140,654	$51,189	$61,354	$261,780
Hampton city	53	301	149	17	810	$134,117	$43,962	$67,363	$269,833
Newport News city	40	281	287	5	1,374	$122,332	$59,341	$37,130	$316,437
Norfolk city	109	168	118	7	528	$168,857	$98,567	$51,507	$590,748
Poquoson city	10	348	150	156	733	$226,740	$58,274	$132,643	$311,350
Portsmouth city	53	296	242	14	1,031	$115,948	$40,479	$54,008	$243,782
Suffolk city	5	1,426	644	545	2,198	$214,610	$35,399	$168,310	$254,324
Virginia Beach city	18	391	210	14	829	$218,105	$81,187	$132,151	$396,073

most a few miles from these homes; however, the model summarized in Equation (4.7) uses properties aggregated at the Census block group level and (most likely) more spatially averaged water quality. These differences are likely to reduce the accuracy of applying Equation (4.7) to estimate benefits.

It is also important to recognize the expected overlap in ecosystem services captured by the hedonic implicit price estimates and the WTP estimates summarized in Section 4.2. In principle, the hedonic price estimate includes residents' values for *all* of the use-related services they receive that depend on water quality. Therefore, in addition to capturing the aesthetic services received by living near the Bay, the hedonic implicit price should include values for recreational services received by near-shore residents. Unfortunately, the hedonic estimates do not provide separate value estimates for these different use-related services. Decomposing the value estimates into separate use-related categories requires additional assumptions, data, or analysis.

4.2.1.5 Nonuse Benefits

Some of the ecosystem services provided by the Chesapeake Bay may be independent of individuals' recreational or other specific uses of the estuary. Measuring values for these nonuse services is more difficult and involves more uncertainty than for recreational and aesthetic services. Nevertheless, several stated-preference studies have estimated water quality values using sample populations that include nonusers. Evidence from these studies indicates that, compared with users of water resources, nonusers have significantly lower but still positive WTP for water quality improvements. Based on this evidence, the following simple benefit transfer equation was specified for estimating nonuse benefits:

$$AggB_{NUt} = N_{NUt} * WTP_{NU}(\Delta WQ_{10t}), \tag{4.9}$$

where

ΔWQ_{10} = change in Chesapeake Bay water quality, expressed on a 10-point rating scale

$AggB_{NUt}$ = aggregate annual benefits in year t (in 2007 dollars) to nonusers of the Chesapeake Bay in Maryland, Virginia, and DC for a specified ΔWQ_{10} increase in water quality

$WTP_{NU}(\Delta WQ_{10})$ = average annual WTP (in 2007 dollars) per nonuser, as a function of the ΔWQ_{10} increase in water quality

N_{NUt} = total number of nonusers in Maryland, Virginia, and DC in year t

To estimate the WTP_{NU} function, results from two meta-analytic studies summarizing evidence from the water quality valuation literature were used. The first, Johnston et al. (2005), included 81 WTP estimates from 34 stated-preference studies. Although these studies addressed a wide variety of water quality changes, for the meta-analysis, they were all converted to a 10-point index (where 0 and 10 represent the worst and best possible water quality, respectively) based on the "RFF water quality ladder" (Vaughan, 1986). The meta-analysis regressed average WTP estimates on water quality measures (baseline and change), characteristics of the water resource and study population, and several study method descriptors. The resulting WTP function can be simplified and summarized as follows:[13]

$$WTP_{NU} = \exp\begin{bmatrix} 2.45 + (0.6827 * \ln(\Delta WQ_{10})) - (0.129 * WQ_{10base}) \\ + (0.005 * INC / CP02) \end{bmatrix} * CP02\,, \qquad (4.10)$$

where

WQ_{10base} = baseline Chesapeake Bay water quality, expressed on the 10-point rating scale

INC = average annual household income of Maryland, Virginia, and DC nonusers in 2007

$CP02$ = price adjustment factor for 2002 to 2007.

The second study, Van Houtven, Powers, and Pattanayak (2007), conducted a similar meta-analysis using a somewhat different sample of studies (18 studies, including 11 for freshwater resources) and WTP estimates (131). A 10-point index based on the RFF ladder was also used to convert water quality changes to a common scale. The resulting WTP function from this study can also be simplified and summarized as follows:[14]

---

[13]The function is a simplified version of the translog unweighted parameter estimate model (Model 2) in Johnston et al. (2005). This model includes several explanatory variables and coefficients, which are summarized in the constant term (2.45). To derive this constant, values were assigned to the other explanatory variables as follows: year is 2007 (*year_index* = 37), study method is a dichotomous choice through a personal interview (*discrete_ch* = 1 and *interview* = 1) to a nonuser-only population (*nonusers* = 1) with a high response rate (*hi_response* = 1), protest and outlier bids are excluded (*protest_bids* = 1 and *outlier_bids* = 1), and the species benefiting from the water quality change are unspecified (*lnWQnon* = *lnwq_change*).

[14]This function is a simplified version of the parsimonious log-linear model in Table 5 in Van Houtven, Powers, and Pattanayak (2007). This model also includes several explanatory variables and coefficients, which are summarized in the constant term (−1.197). To derive this constant in a way that is consistent with the previous function, values were assigned to the other explanatory variables as follows: year is 2007 (*studyyr73* = 34), study method is a personal interview (*inperson* = 1) to a nonuser-only population (*pctuser* = 0) with a high response rate (*responserate* = 100), publication outlet is peer reviewed (*dpubjrlbk* = 1), and the water quality change is not expressed in terms of recreational uses (*lnwq10chru* = 0).

$$WTP_{NU} = \exp\begin{bmatrix} -1.197 + (0.823 * \ln(\Delta WQ_{10})) + (0.0801 * WQ_{10base}) \\ + (0.8969 * \ln(INC / CP00)) \end{bmatrix} * CP00 , \qquad (4.11)$$

where

$CP00$ = price adjustment factor for 2000 to 2007

Using these functions, WTP_{NU} can be estimated for selected values of ΔWQ_{10}, WQ_{10base}, and *INC*. For *INC*, U.S. average household income in 2007 of $67,610 was used (U.S. Census Bureau, 2008a). As an example, based on these inputs, WTP_{NU} for a 2-unit change in water quality is estimated to be $16.33 using the Johnston et al. (2005) function and $27.75 using the Van Houtven, Powers, and Pattanayak (2007) function.

Estimates of the percentage of Maryland, Virginia, and DC residents who are nonusers of the Chesapeake Bay are not readily available; however, they can be roughly approximated from recreational participation statistics for the area. For example, data from the 2006 Virginia Outdoors Survey suggest that (1) 92% of households in Virginia did not take any beach trips to the Chesapeake Bay, (2) 84% did not engage in saltwater fishing, and (3) 92% did not engage in powerboating. Assuming that these proportions represent independent probabilities of nonuse, then the combined probability (proportion) of nonuse for these primary activities is roughly 70%. Applying this percentage to the Maryland, Virginia, and DC population in 2007, which was 13,918,727, suggests that the number of nonusers (N_{NU}) is approximately 9,743,109. Projected state-level populations for future years can be used in a similar way to estimate future values of N_{NU}.

<u>4.2.1.5.1 Limitations and Uncertainties</u>

As with the recreational boating services model described in Section 4.2.1.2, one of the main practical limitations of applying these meta-analysis models is the water quality index used. Translating changes in estimated water quality to the WQ_{10} metric requires strong assumptions. Another inherent limitation of using the meta-analytic models as benefit transfer functions is their lack of sensitivity to the spatial scale of water quality changes.

In addition to the limitations that primarily contribute uncertainty in the WTP_{NU} estimates, there is also significant uncertainty associated with the measurement of N_{NU}. First, defining criteria for distinguishing users and nonusers of the Bay is somewhat inherently subjective. Second, statistics on overall rates of visitation and use of the Bay by Maryland, Virginia, and DC households are not readily available.

A final caveat for this approach to estimating nonuse values for water quality improvements in the Bay is that, by design, it only includes nonuse values for *nonusers*. However, it is not unreasonable to suspect that users also benefit to some extent from nonuse services from the Bay. Whereas these types of nonuse values are likely to be captured in, for example, the Lipton (2004) WTP values for boaters used in Equation (4.5), they are not included in the benefit estimates in Equations (4.4), (4.6), and (4.7) for recreational anglers, beach users, and residents, respectively.

4.2.2 Neuse River and Albemarle Pamlico Sound Estuaries

This section describes transfer models for assessing the commercial and recreational benefits of reduced N loadings to the Neuse River Estuary. In contrast to the Chesapeake Bay benefit transfer models, these models use changes in N loadings to the estuary, rather than changes in estuarine water quality, as model inputs.

4.2.2.1 Commercial Fishing Benefits: the Blue Crab Fishery

As discussed in Section 4.1.1, few examples of empirical bioeconomic models link changes in nutrient-related water quality to changes in productivity of commercial fisheries; however, one exception is a study by Smith (2007). This study, which is applied to the Neuse River estuary, estimated the dynamic effects of a 30% reduction in N loads to the estuary on blue crab stocks, commercial catch levels, and the producer and consumer surplus derived from this fishery.

Smith (2007) applied a two-patch predator-prey model that incorporated both direct and indirect effects of hypoxia (i.e., low DO) on blue crab communities. Direct effects include the movement of blue crab to water habitats with higher DO content. Indirect effects include the dying off of blue crab prey. The model compares producer and consumer surplus changes under the existing open-access institutional structure to a 30% reduction of N loadings in the same structure. The model was parameterized using results and estimates derived from several other studies. To address uncertainty, the values of three key parameters—economic speed of adjustment under open-access conditions, biological spatial connectivity, and price elasticity of demand—were each allowed to take on three different values. For a 30% reduction in N loadings to the estuary, the present value (100-year time horizon and 4.5% discount rate) of producer benefits ranged from $0.7 million to $5.9 million (in 2002 dollars), and the present value of consumer surplus ranged from $3.15 million to $425.20 million. The combined present value of producer and consumer surplus changes was estimated to range from $3.8 to $31.0 million.

To estimate the annual aggregate benefits from the blue crab fishery due to different percentage reductions in N loads, the results reported in Smith (2007) can be rescaled (i.e., to \$130,000 to \$1.03 million per 1% reduction in N loadings), annualized, and converted to 2007 dollars using the CPI.

4.2.2.1.1 Limitations and Uncertainties

The large range of the benefit estimates reported above reflects uncertainty in three key model parameters—economic speed of adjustment under open-access conditions, biological spatial connectivity, and price elasticity of demand. However, the model includes at least 16 other parameters whose values are drawn from other studies; thus, the overall uncertainty in these benefit estimates is most likely understated by this range.

In addition, by simply rescaling the results reported in Smith (2007) to address changes other than a 30% reduction in N loads, it must be assumed that benefits are directly proportional to the percentage reduction in N loads. Because of the inherent nonlinearities in the bioeconomic model, this is a very strong assumption, particularly for changes that are much smaller or larger than 30%.

4.2.2.2 Recreational Fishing Services

To estimate the benefits from improvements in recreational fishing services due to reductions in N loadings to the Neuse, a benefit transfer model originally developed to assess the nutrient-reduction benefits of EPA's effluent guidelines for Consolidated Animal Feeding Operations (CAFOs) (EPA, 2002) can be applied. For that analysis, EPA conducted a case study evaluating the potential economic benefits of a reduction in nutrient loadings via changes in recreational fishing opportunities in North Carolina's Albemarle and Pamlico Sounds (APS) estuary (Van Houtven and Sommer, 2002). The Neuse River estuary is a subestuary within the APS system.

To estimate the value of reductions in N loads, the APS case study relied on economic value estimates obtained from two related studies—Kaoru (1995) and Kaoru, Smith, and Liu (1995). Both studies used recreational data obtained from a 1981 to 1982 intercept survey of recreational fishermen conducted at 35 boat ramps or marinas within the APS estuary.

Kaoru (1995) used a three-level nested random utility model (RUM), which broke the recreational fishing decision into three stages: a decision on the duration of the trip (1, 2, 3, or more than 3 days), a decision on which of the five regions to visit, and a decision on which of the individual sites within the region to visit. The impact of N (and phosphorus) loadings was specifically investigated in the second stage of the decision process (regional choice). A 25% reduction in N loadings for the entire APS estuary resulted in a benefit estimate of $4.70 (in 1982 dollars) per person-trip.

Kaoru, Smith, and Liu (1995) also used a RUM approach to estimate the value of improving water quality. First, a household production function (HPF) was estimated to predict expected catch rates for individuals based on variables such as equipment used; effort exerted; and the physical characteristics of the fishing site, including pollutant loadings. Second, the HPF model was used to predict the impact of a 36% reduction in N loadings on expected catch rates. The estimated values ranged from $0.76 to $6.52 (in 1982 dollars) per person-trip.

Based on a systematic review of the value estimates reported in these studies, the CAFO case study selected three estimates to include in the benefit transfer model—$4.70 per person-trip for a 25% reduction in N loads (Kaoru, 1995) and $3.95 and $6.52 per person for a 36% reduction (Kaoru, Smith, and Liu, 1995).

To apply these estimates, they were converted to comparable units. First, they were converted to 2007 dollars using the CPI. Second, they were rescaled to values per 1% reduction in loadings (i.e., dividing by 25 and 36, respectively). The resulting three unit values are $0.40, $0.24, and $0.39 per person-trip per 1% reduction in N loads to the APS.

A further adjustment is necessary to convert these values into per-ton units. According to Kaoru (1995), the average N load to the APS estuary at the time the study was conducted was 1,741 tons per bordering county per year, which translates to a total of 22,633 tons of N loadings per year because of the 13 counties bordering the APS estuary in North Carolina. The resulting three unit values are $0.0018, $0.0010, and $0.0017 per person-trip per 1-ton reduction in N loads to the APS.

To estimate the aggregate annual recreational fishing benefits of total reductions in N loads to the APS estuary, the following benefit transfer equation was specified:

$$AggB_{APSfisht} = V \times \Delta L_t \times T_t, \tag{4.12}$$

where

$AggB_{APSfisht}$	=	the aggregate annual recreational fishing benefits from reductions in N loads to the APS estuary in year t (in 2007 dollars)
V	=	the annual per trip value per-unit (either in tons per year or percentage) reduction in N (in 2007 dollars)

ΔL_t = reduction in N loadings (either in tons per year or percentage) to the APS estuary in year t

T_t = the total number of annual fishing trips to the APS estuary (person-trips per year) in year t

Although the unit value (V) estimates derived from Kaoru (1995) and Kaoru, Smith, and Liu (1995) are based on data only for boating anglers, it was assumed that they apply to *all* recreational fishing trips (T) in the APS. Data on visitation rates for recreational anglers in the APS estuary are available from the MRFSS, which contains information on the number, type, and destination of recreational fishers for several coastal regions in the United States. For 2006, the MRFSS data provide an estimate of 753,893 person-trips to the APS for recreational fishing. To estimate trips in year t, this estimate can be augmented by the projected population growth rate in eastern North Carolina from 2006 to year t.

4.2.2.2.2 Limitations and Uncertainties

The following limitations and uncertainties should be considered when interpreting these recreational fishing benefit estimates. First, the value estimates are based on fishing activity data that are more than 2 decades old. The analysis assumes that the benefits of water quality changes have remained constant (in real terms) over this period.

Second, the value estimates obtained from the two existing studies were based on percentage reductions in nutrients that were uniform across the APS estuary. By converting these estimates into per-ton terms and applying them only to the Neuse River N load reductions, the analysis implicitly assumes that average per-trip benefits do not vary with respect to the spatial distribution of the loadings reductions.

Third, the original value estimates are based on data only from boat fishermen; however, the analysis assumes that these values are appropriate for both boat and nonboat fishers.

4.3 Summary of Modeling Framework

Figures 4-3 and 4-4 summarize the main proposed modeling steps for assessing the benefits of reductions in N loadings to the Chesapeake Bay and APS/Neuse estuaries, respectively. Figure 4-3, in particular, highlights the key modeling gap that exists in the Chesapeake Bay (i.e., quantifying the relationship between N loadings to the Bay and changes in Bay water quality).

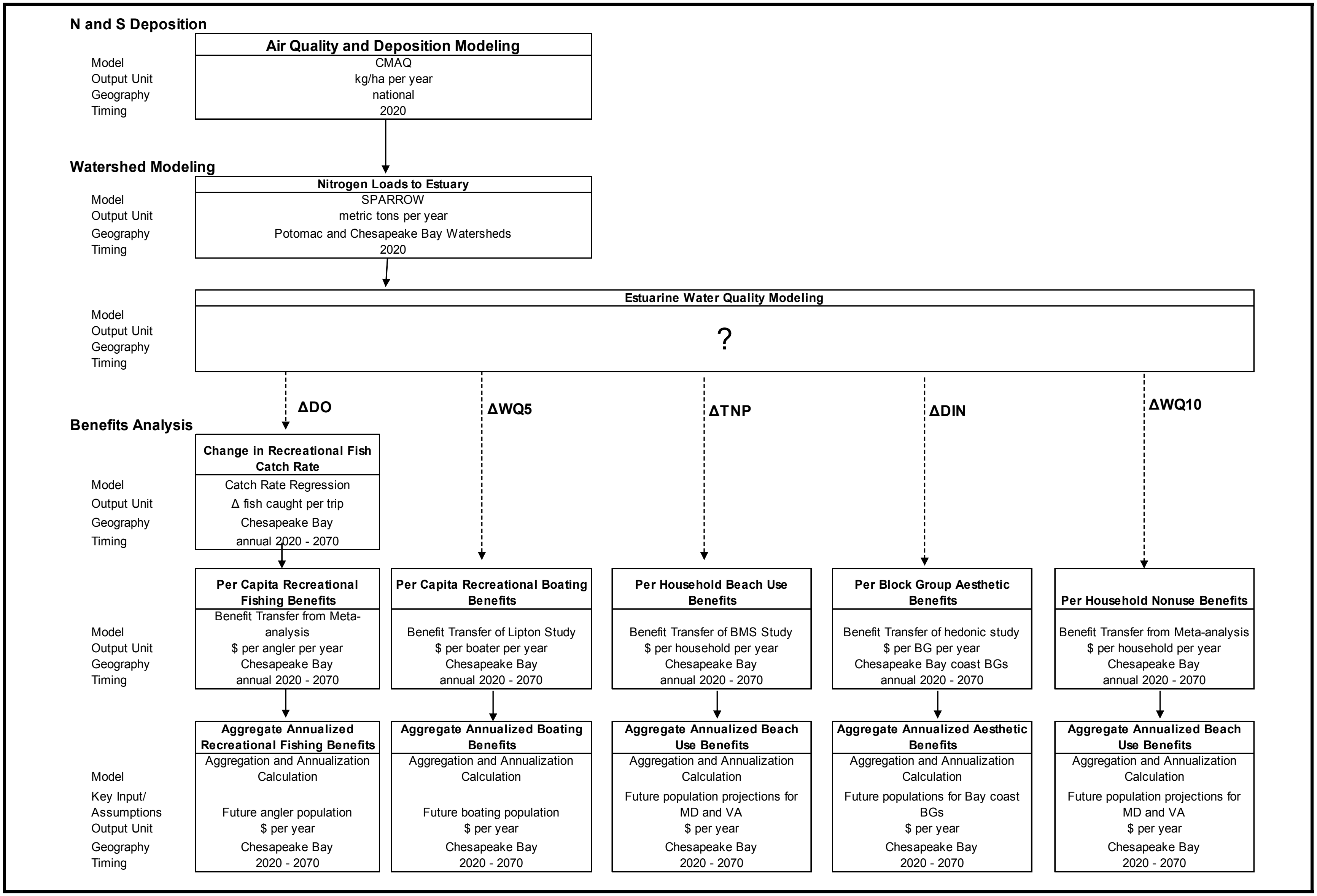

Figure 4-3. Key Modeling Steps for Assessing the Benefits of Reduced N Loadings to the Chesapeake Bay Estuary

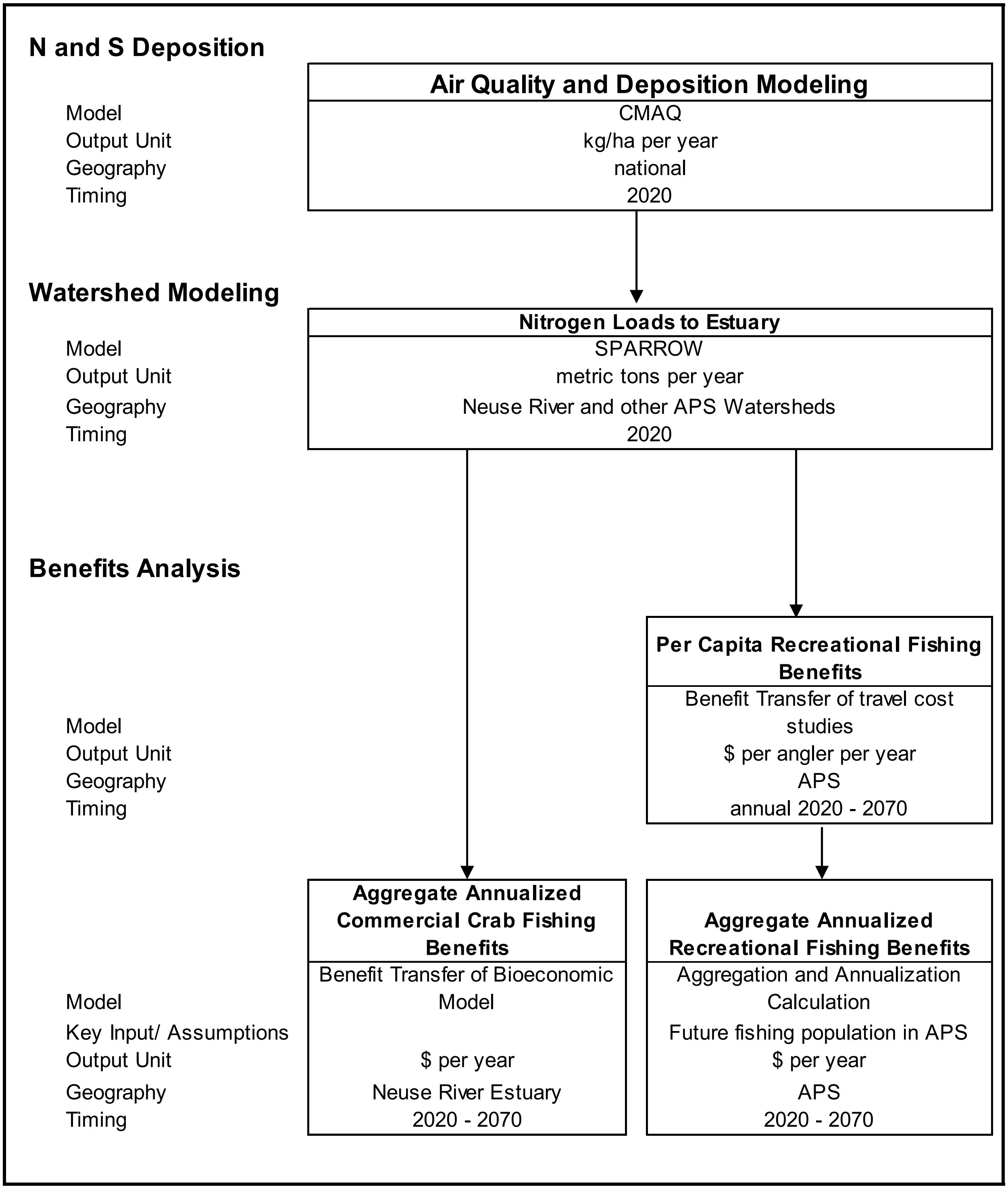

Figure 4-4. Key Modeling Steps for Assessing the Benefits of Reduced N Loadings to the Neuse/APS Estuaries

4.4 References

Anderson, E. 1989. "Economic Benefits of Habitat Restoration: Seagrass and the Virginia Hard-Shell Blue Crab Fishery." *North American Journal of Fisheries Management* 9:140-149.

Bartik, T.J. 1988. "Measuring the Benefits of Amenity Improvements in Hedonic Price Models." *Land Economics* 64(2):172-183.

Bockstael, N.E., K.E. McConnell, and I.E. Strand. 1988. "Benefits from Improvements in Chesapeake Bay Water Quality, Volume III." Environmental Protection Agency Cooperative Agreement CR-811043-01-0.

Bockstael, N.E., K.E. McConnell, and I.E. Strand. 1989. "Measuring the Benefits of Improvements in Water Quality: The Chesapeake Bay." *Marine Resource Economics* 6:1-18.

Bricker, S., D. Lipton, A. Mason, M. Dionne, D. Keeley, C. Krahforst, J. Latimer, and J. Pennock. 2006. *Improving Methods and Indicators for Evaluating Coastal Water Eutrophication: A Pilot Study in the Gulf of Maine*. NOAA Technical Memorandum NOS NCCOS 20. Silver Spring, MD: National Ocean Service, National Centers for Coastal Ocean Science, Center for Coastal Monitoring and Assessment.

Bricker, S., B. Longstaff, W. Dennison, A. Jones, K. Boicourt, C. Wicks, and J. Woerner. 2007. *Effects of Nutrient Enrichment in the Nation's Estuaries: A Decade of Change.* NOAA Coastal Ocean Program Decision Analysis Series No. 26. Silver Spring, MD: National Centers for Coastal Ocean Science.

Bricker, S.B., J.G. Ferreira, and T. Simas. 2003. "An Integrated Methodology for Assessment of Estuarine Trophic Status." *Ecological Modelling* 169:39-60.

Chesapeake Bay Program. n.d. "Chesapeake Bay Program: A Watershed Partnership." Available at http://www.chesapeakebay.net/.

Chesapeake Bay Program. "Bay Grass Abundance (Baywide)." Available at http://www.chesapeakebay.net/ status_baygrasses.aspx?menuitem=19652. Last updated May 21, 2008.

Environmental Systems Research Institute, Inc. (ESRI). 2002. "Census Block Groups: DC, DE, MD, and VA." [CD-ROM]. *ESRI Data & Maps 2002*.

Johnston, R.J., E.Y. Besedin, R. Iovanna, C.J. Miller, R.F. Wardwell, and M.H. Ranson. 2005. "Systematic Variation in Willingness to Pay for Aquatic Resource Improvements and Implications for Benefit Transfer: A Meta-Analysis." *Canadian Journal of Agricultural Economics* 53:221-248.

Johnston, R.J., M.H. Ranson, E.Y. Besedin, and E.C. Helm. 2006. "What Determines Willingness to Pay per Fish? A Meta-Analysis of Recreational Fishing Values." *Marine Resource Economics* 21:1-32.

Kahn, J.R., and W.M. Kemp. 1985. "Economic Losses Associated with the Degradation of an Ecosystem: The Case of Submerged Aquatic Vegetation in Chesapeake Bay." *Journal of Environmental Economics and Management* 12:246-263.

Kaoru, Y. 1995. "Measuring Marine Recreation Benefits of Water Quality Improvements by the Nested Random Utility Model." *Resource and Energy Economics* 17(2):119-136.

Kaoru, Y., V. Smith, and J.L. Liu. 1995. "Using Random Utility Models to Estimate the Recreational Value of Estuarine Resources." *American Journal of Agricultural Economics* 77:141-151.

Kaval, P., and J. Loomis. 2003. *Updated Outdoor Recreation Use Values with Emphasis on National Park Recreation*. Final Report October 2003, under Cooperative Agreement CA 1200-99-009, Project number IMDE-02-0070.

Knowler, D. 2002. "A Review of Selected Bioeconomic Models: With Environmental Influences in Fisheries." *Journal of Bioeconomics* 4:163-181.

Krupnick, A. 1988. "Reducing Bay Nutrients: An Economic Perspective." *Maryland Law Review* 47(2):453-480.

Leeworthy, V.R., and P.C. Wiley. 2001. *Current Participation Patterns in Marine Recreation*. Silver Spring, MD: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Ocean Service, Special Projects.

Lipton, D. W. 1999. "Pfiesteria's Economic Impact on Seafood Industry Sales and Recreational Fishing." In *Proceedings of the Conference, Economics of Policy Options for Nutrient Management and Pfiesteria*. B. L. Gardner and L. Koch, eds., pp. 35–38. College Park, MD: Center for Agricultural and Natural Resource Policy, University of Maryland.

Lipton, D. 2004. "The Value of Improved Water Quality to Chesapeake Bay Boaters." *Marine Resource Economics* 19:265-270.

Lipton, D. 2006. "Economic Impact of Maryland Boating in 2005." University of Maryland Sea Grant Extension Program. Available at ftp://ftp.mdsg.umd.edu/Public/MDSG/rec_boat05.pdf.

Lipton, D. 2008. "Economic Impact of Maryland Boating in 2007." University of Maryland Sea Grant Extension Program. Available at ftp://ftp.mdsg.umd.edu/Public/MDSG/rec_boat07.pdf.

Lipton, D.W., and R.W. Hicks. 1999. "Linking Water Quality Improvements to Recreational Fishing Values: The Case of Chesapeake Bay Striped Bass." In *Evaluating the Benefits of Recreational Fisheries. Fisheries Centre Research Reports* 7(2), T.J. Pitcher, ed., pp. 105-110. Vancouver: University of British Columbia.

Lipton, D.W., and R.W. Hicks. 2003. "The Cost of Stress: Low Dissolved Oxygen and the Economic Benefits of Recreational Striped Bass Fishing in the Patuxent River." *Estuaries* 26(2A):310-315.

Massey, M., S. Newbold, and B. Gentner. 2006. "Valuing Water Quality Changes Using a Bioeconomic Model of a Coastal Recreational Fishery." *Journal of Environmental Economics and Management* 52:482-500.

Millennium Ecosystem Assessment (MEA). 2005. Ecosystems and Human Well-being: Wetlands and Water. Synthesis. A Report of the Millennium Ecosystem Assessment. Washington, DC: World Resources Institute.

Mistiaen, J.A., I.E. Strand, and D. Lipton. 2003. "Effects of Environmental Stress on Blue Crab (*Callinectes sapidus*) Harvests in Chesapeake Bay Tributaries." *Estuaries* 26(2a):316-322.

Morgan, C., and N. Owens. 2001. "Benefits of Water Quality Policies: The Chesapeake Bay." *Ecological Economics* 39:271-284.

Murray, T., and J. Lucy. 1981. "Recreational Boating in Virginia: A Preliminary Analysis." Special Report in Applied Marine Science and Ocean Engineering No. 251. Available at http://www.vims.edu/GreyLit/VIMS/sramsoe251.pdf.

National Marine Manufacturers Association (NMMA). 2008. "2007 Recreational Boating Statistical Abstract." Available at http://www.nmma.org/facts/boatingstats/2007/files/Abstract.pdf.

National Oceanic and Atmospheric Administration (NOAA). (2007, August). "Annual Commercial Landing Statistics." Available at http://www.st.nmfs.noaa.gov/st1/commercial/landings/annual_landings.html.

National Oceanic and Atmospheric Administration (NOAA). "Recreational Fisheries." Available at www.st.nmfs.noaa.gov/st1/recreational/overview/overview.html. Last updated February 12, 2009.

Parsons, G., A.O. Morgan, J.C. Whitehead, and T.C. Haab. 2006. "The Welfare Effects of Pfiesteria-Related Fish Kills: A Contingent Behavior Analysis of Seafood Consumers." *Agricultural and Resource Economics Review* 35(2):1-9.

Poor, P.J., K.L. Pessagno, and R.W. Paul. 2007. "Exploring the Hedonic Value of Ambient Water Quality: A Local Watershed-Based Study." *Ecological Economics* 60:797-806.

Smith, M.D. 2007. "Generating Value in Habitat-Dependent Fisheries: The Importance of Fishery Management Institutions." *Land Economics* 83(1):59-73.

Sweeney, J. 2007. "Impacts of CAMD 2020 CAIR on Nitrogen Loads to the Chesapeake Bay." University of Maryland, Chesapeake Bay Program Office.

U.S. Census Bureau. 2008a. “Annual Social and Economic (ASEC) Supplement.” Available at http://pubdb3.census.gov/macro/032008/hhinc/new02_001.htm.

U.S. Census Bureau. 2008b. “Housing Unit Estimates for Counties of MD and VA: April 1/2000 to July 1/2007.” Available at http://www.census.gov/popest/housing/HU-EST2007-CO.html.

U.S. Environmental Protection Agency (EPA). December 2002. *Environmental and Economic Benefit Analysis of Final Revisions to the National Pollutant Discharge Elimination System Regulation and the Effluent Guidelines for Concentrated Animal Feeding Operations*. EPA-821-R-03-003. Washington, DC: U.S. Environmental Protection Agency, Office of Water, Office of Science and Technology.

U.S. Environmental Protection Agency (EPA). 2009. *Risk and Exposure Assessment for Review of the Secondary National Ambient Air Quality Standards for Oxides of Nitrogen and Oxides of Sulfur.* Research Triangle Park, NC: U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards. EPA-452/R-09-008a.

Valigura, R.A., R.B. Alexander, M.S. Castro, T.P. Meyers, H.W. Paerl, P.E. Stacy, and R.E. Turner. 2001. *Nitrogen Loading in Coastal Water Bodies: An Atmospheric Perspective*. Washington, DC: American Geophysical Union.

Van Houtven, G. and A. Sommer. December 2002. *Recreational Fishing Benefits: A Case Study of Reductions in Nutrient Loads to the Albemarle-Pamlico Sounds Estuary. Final Report.* Prepared for the U.S. Environmental Protection Agency. Research Triangle Park, NC: RTI International.

Van Houtven, G.L., J. Powers, and S.K. Pattanayak. 2007. “Valuing Water Quality Improvements Using Meta-Analysis: Is the Glass Half-Full or Half-Empty for National Policy Analysis?” *Resource and Energy Economics* 29:206-228.

Vaughan, W.J. 1986. “The Water Quality Ladder.” *The Use of Contingent Valuation Data for Benefit/Cost Analysis in Water Pollution Control*. In R.C. Mitchell, and R.T. Carson, eds. CR-810224-02. Prepared for the U.S. Environmental Protection Agency, Office of Policy, Planning, and Evaluation.

Virginia Department of Conservation and Recreation. 2007. “2006 Virginia Outdoors Survey.” Available at http://www.dcr.virginia.gov/recreational_planning/documents/vopsurvey06.pdf.

Whitehead, J.C., T.C. Haab, and G.R. Parsons. 2003. “Economic Effects of Pfiesteria.” *Ocean & Coastal Management* 46(9-10):845-858.

SECTION 5
TERRESTRIAL ENRICHMENT

5.1 Overview of Affected Ecosystems and Ecological Endpoints

Terrestrial enrichment occurs when terrestrial ecosystems receive N loadings in excess of natural background levels, either through atmospheric deposition or direct application. Evidence presented in the Integrated Science Assessment (EPA, 2008) supports a causal relationship between atmospheric N deposition and biogeochemical cycling and fluxes of N and carbon in terrestrial systems. Furthermore, evidence summarized in the report supports a causal link between atmospheric N deposition and changes in the types and number of species and biodiversity in terrestrial systems.

Figure 5-1 provides a conceptual model that traces the effects of excess N deposition in terrestrial ecosystems to the main ecosystem outcomes and affected services. The relative importance and magnitudes of these linkages depend in part on the type of terrestrial ecosystem affected. Three ecosystems considered to be particularly vulnerable to these enrichment effects are coastal sage scrub (CSS), mixed conifer forests (MCF), and western grasslands.

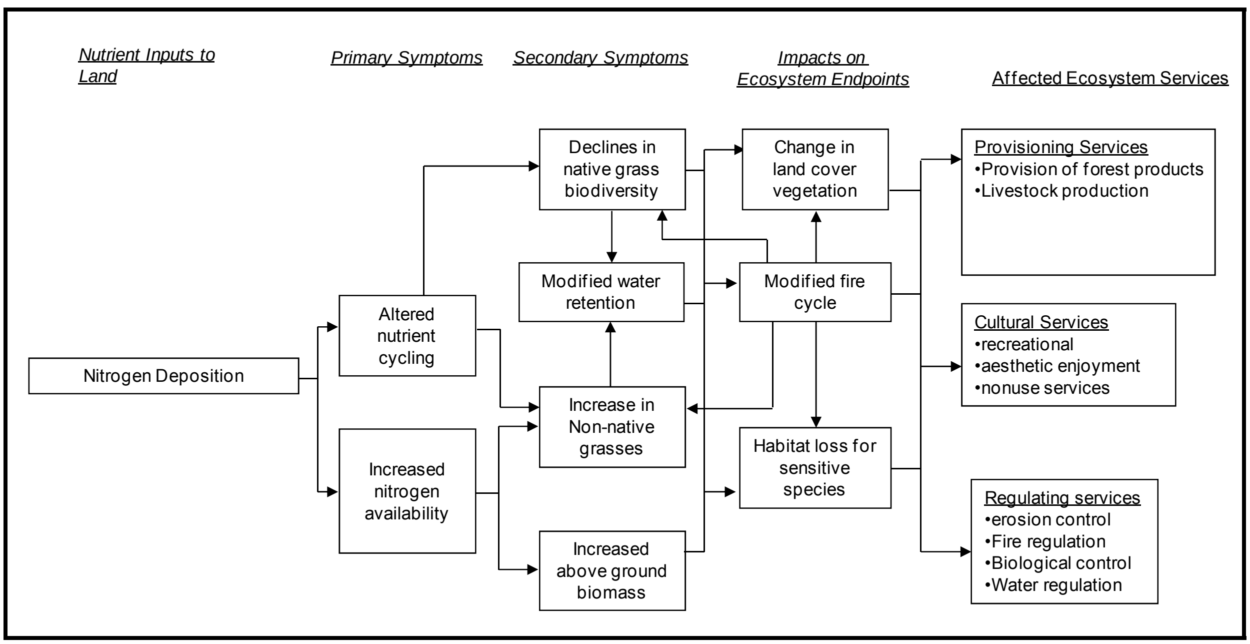

Figure 5-1. Conceptual Diagram of Ecosystem Service Impairments Associated with Terrestrial Nutrient Enrichment

5.1.1 Coastal Sage Scrub

The range of CSS extends from north of San Francisco down to Baja California in the lower elevation coastal range of California (see Figure 5-2). It consists of more than 50 aromatic shrub and subshrub species; however, the species composition may vary with location (Westman, 1981b). CSS is considered a fire-adapted community, meaning that although vegetation layers may be destroyed in fires, CSS soil seed banks can withstand fire and, in some species, require fire to open the seed cases. However, many CSS species can flourish and propagate in the absence of any fire (Keeler-Wolf, 1995).

Figure 5-2. Range of Coastal Sage Scrub Ecosystems

CSS is subject to several pressures, such as land conversion, grazing, fire, and pollution, all of which have been observed to induce declines in other ecosystems (Allen et al., 1998). Research suggests that both fire and increased N can enhance the growth of nonnative grasses in established CSS ecosystems. It is hypothesized that many stands are no longer limited by N and have instead become N-saturated due to atmospheric N deposition (Allen et al., 1998; Westman, 1981a). N availability may favor the germination and growth of nonnative grasses, which can create a dense network of shallow roots that slow the diffusion of water through soil, decrease the percolation depth of precipitation, and decrease the water storage capability of the soil and underlying bedrock (Wood et al., 2006). Establishment of CSS species may be decreased because of decreased water and N availability at depths where more woody CSS tap roots are found (Keeler-Wolf, 1995; Wood et al., 2006).

CSS has been declining in land area and in shrub density for the past 60 years and in many places is being replaced by nonnative annual grasses (Allen et al., 1998; Padgett and Allen, 1999). N deposition has been suggested as a possible cause or factor in this ecosystem alteration (EPA, 2008).

5.1.2 Mixed Conifer Forest

Figure 5-3 illustrates the range of MCF in California. Ponderosa pine (*Pinus ponderosa*), white fir (*Abies concolor*), sugar pine (*P. lambertiana*), and incense cedar (*Calocedrus decurrens*) are the predominant species on moist windward slopes, whereas Jeffrey pine (*P. jeffreyi*) and white fir are commonly found on leeward slopes and at higher elevations in the mixed conifer elevation range. Important deciduous components of the MCF are canyon live oak (*Quercus chrysolepis*), black oak (*Quercus kelloggi*), and quaking aspen (*Popus tremuloides*). These stands support a number of shrubs, subshrubs, and annual and perennial forbs, as well as mountain meadows (Minnich, 2007).

At the individual tree level, elevated atmospheric N can shift the ratio of above-ground to below-ground biomass. Elevated pollution levels may result in increased uptake of nutrients via the canopy, decreased N intake requirements on root structures, and increased demand for carbon dioxide (CO_2) uptake and photosynthetic structures to maintain the carbon balances. Therefore, the increased nutrient availability stimulates above-ground growth and increases foliar production, while reducing the demand for below-ground nutrient uptake (Fenn et al., 2000).

At the stand level, elevated atmospheric N has been associated with increased stand density, although other factors, such as fire suppression, also contribute to increased density and can increase mortality rates (EPA, 2008). As older trees die, they are replaced with younger, smaller trees. Smaller trees allow more sunlight through the canopy and, combined with an increased availability of N, may allow for more trees to be established. Increased stand densities with younger-age classes are observed in the San Bernardino Mountains, where air pollution levels are among the highest found in the California MCF ranges studied (Minnich et al., 1995;

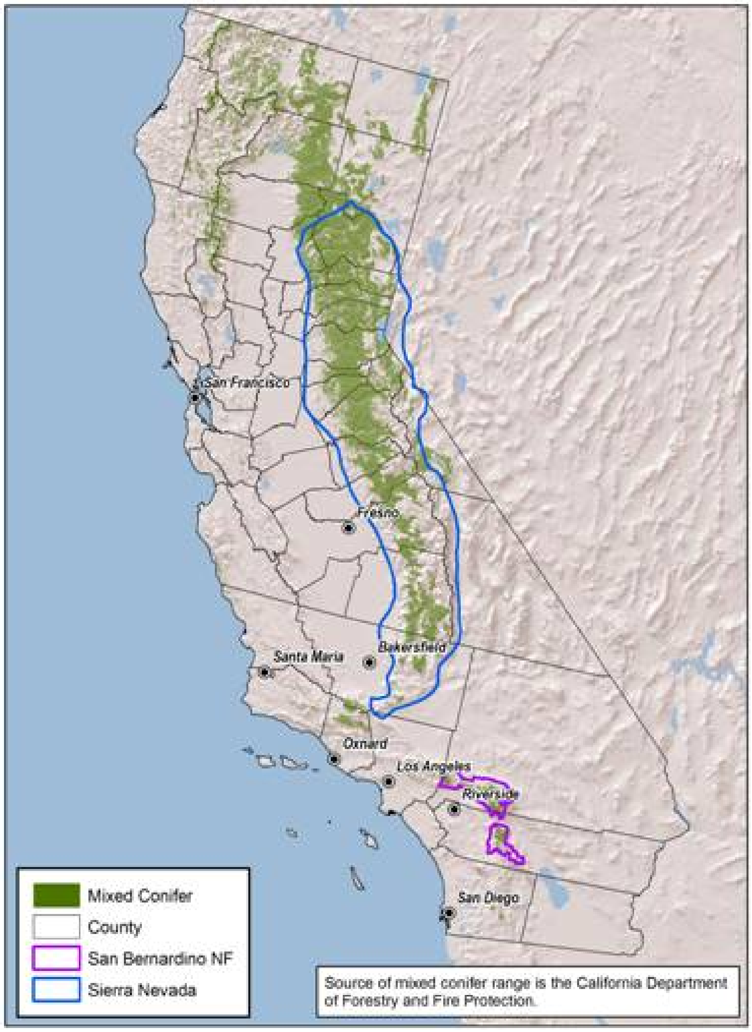

Figure 5-3. Range of California's Mixed Conifer Forests

Fenn et al., 2008). These shifts in stand density and age distribution result in vegetation structure shifts which, in turn, may affect population and community dynamics of understory plants and animals, including threatened and endangered species.

High concentrations of O_3 and atmospheric N can generate increased needle and branch turnover. Needle turnover significantly increases litterfall. Litter biomass has been observed to increase in areas with elevated atmospheric N deposition up to 15 times more than in areas with low deposition, and the litter is seen to have higher concentrations of N (Fenn et al., 2000; Grulke et al., 2008). Elevated litter N levels may facilitate faster rates of microbial decomposition initially, but over the long term high N levels slow litter decomposition, and litter accumulates on the forest floor (Grulke et al., 2008; EPA, 2008). The increased litter depth may then affect subcanopy growth and stand regeneration over long periods of time.

At the highest levels of N deposition, native understory species were seen to decline (Allen et al., 2007). In addition to this decline in native understory diversity, changes in decreased fine-root mass, increased needle turnover, and the associated chemostructural alterations, MCF that are exposed to elevated pollutant levels have an increasing susceptibility to drought and beetle attack (Grulke et al., 1998, 2001; Takemoto et al., 2001). These stressors often result in the death of trees, producing an increased risk of wildfires.

5.1.3 Western Grasslands

Roughly one-half of the contiguous 48 states were once covered by grasslands, and a large majority of these were located west of the Mississippi River. Nearly 600 million acres of these grasslands were located in the Central Plains region between the Mississippi River and the Rocky Mountains, and over 300 million more acres were found west of the Rocky Mountains. Mainly because of the conversion of these lands to croplands, the current coverage of grasslands west of the Mississippi is about 400 million acres (Conner et al., 2001). Currently, almost all of the remaining grasslands in the Central Plains are privately owned, whereas west of the Rockies they are predominantly under federal ownership.

There is growing evidence that existing grassland ecosystems in the western United States are being altered by elevated levels of N inputs, including inputs from atmospheric deposition. Grasslands in the western United States are typically N-limited ecosystems dominated by a diverse mix of perennial forbs and grass species (Clark and Tilman, 2008; Suding et al., 2005). Additional N may affect species diversity and composition, as well as contribute to altered nutrient cycles, fire regimes, and erosion susceptibility. The productivity of grasslands may change, as well as their suitability for forage. These changes may lead to less productive grasslands and or the presence of increased numbers of nonnative species.

The different species of plants in grasslands do not respond identically to increased N and/or other resources. For example, the response of N-fixing forbs to increased N was observed to be lower than non-N-fixing forbs (Suding et al., 2005). Elevated atmospheric N deposition may reduce the N limitations on these ecosystems, affecting species abundance and composition. In general, research indicates that when N deposition is increased both a loss of species richness and a change in species composition may occur (Monaco et al., 2003; Stevens et al., 2009; Allen et al., 2009; Schwinning et al., 2005). Clark and Tilman (2008) observed a 17% reduction in species number when N was added to grasslands and rare species were lost at a higher ratio (26%) compared to common species (8 to 16%) (Suding et al., 2005; Clark and Tilman, 2008).

Species composition also appears to respond to increased levels of N. A shift in species competition was seen in a long-term deposition study, where N-efficient species were gradually dominated by more N-inefficient species (Clark and Tilman, 2008). In several grassland ecosystems, reduced species diversity is also accompanied by an increase in non-native, invasive species (Allen et al., 2009; Monaco et al., 2003; Schwinning et al., 2005). A comparison of nonnative annual species versus native perennial species in Utah grasslands consistently observed a higher root biomass and a lower above-ground carbon-to-N ratio in native perennials under all tested N concentrations, suggesting that perennials are more competitively structured for N-limited ecosystems (Monaco et al., 2003).

Invasive annuals tend to respond more readily and maintain high growth rates with increased N when compared to native perennials, which tend to respond conservatively to increased N (Monaco et al., 2003; Schwinning et al., 2005). For example, in experimental plots with added N, nonnative grasses in Joshua Tree National park germinated earlier and produced more seeds than native species (Allen et al., 2009). In contrast, seed production was greatly reduced in nonnative species when low N was maintained over a number of years in an intermountain western grassland ecosystem (Monaco et al., 2003).

The seasonal timing of N deposition and availability is also a factor. Both atmospheric deposition and vegetation growth experience a seasonal component. Depending on the primary growth season, species that respond more quickly and favorably to N may be able to take advantage of seasonal pulses (Schwinning et al., 2005). These species are generally nonnative annual grasses, which exhibit more opportunistic responses to N.

These research efforts point to a possible relation between the enhanced growth of nonnatives, especially in habitats where N is elevated, and the reduced abundance and diversity of native species.

Research indicates that differing fire regimes and changes to soil biochemistry along with elevated N levels may also be important contributing factors that both respond to and promote changes in vegetation. Fire can greatly alter the biomass and water and nutrient conditions in an ecosystem depending on frequency, intensity, and timing. Species composition changes may change fire regimes, as the biological characteristics of certain nonnative species may result in more intense or frequent fires and increased competition for water resources (Finnoff et al., 2008). It should be noted that not all nonnative species increase fire occurrence as noted by Grace et al. (2001). The increased grass biomass associated with increased N resources has been cited as contributing to an increase in fire frequency (Allen et al., 2009). There is also some evidence that changes in biomass and species may affect fire intensity, fire frequency, and increased soil erosion (Rao et al., 2009; Finnoff et al., 2008; Monaco et al., 2003; Allen et al., 2009).

The characteristics intrinsic to the soil itself can alter available N and fine, shallow root systems (Allen et al., 2009). Additionally, Rao et al. (2009) suggested that increases in nonnative grasses may alter below-ground concentrations of carbon, further changing the response of the vegetation in the soils. Nonnative species tend to allocate fewer resources to root structure, which can result in decreased soil stability and increased erosion (Finnoff et al., 2008; Monaco et al., 2003). Erosion may result in a variable loss of nutrients from the ecosystem.

Management practices for the ecosystem, such as grazing, may positively or negatively affect nutrient availability and biomass (Finnoff et al., 2008). Moderate grazing may help favor native grasses by maintaining competition for N; however, intense grazing can increase nutrient cycling and availability, favoring a nonnative shift (Weiss, 1999).

There are multiple possible feedbacks between increased N and ecological effects, including changes to soil chemistry, enhanced growth of nonnative grasses, decreased biodiversity, increased grass biomass, increased fuel loads and more intense and frequent fires, and diminished abundance and diversity of native species. Although these species shifts in diversity and composition are well documented, quantification and modeling of the effects of N on the grassland ecosystem are ill defined. The combination of multiple, subtle, compounding, and dependent ecosystem effects introduces high levels of uncertainty into proposed models.

5.2 Overview of Affected Ecosystem Services

5.2.1 Effects on Provisioning Services

One of the main provisioning services potentially affected by N deposition is grazing opportunities offered by western grasslands for livestock production. Although N deposition on these grasslands can offer supplementary nutritive value and promote overall grass production, there are concerns that fertilization may favor invasive grasses and shift the species composition away from native grasses. This process may ultimately reduce the productivity of grasslands for livestock production.

Losses due to invasive grasses can be significant; for example, based on a bioeconomic model of cattle grazing in the upper Great Plains, Leitch, Leistritz, and Bangsund (1996) and Leistritz, Bangsund, and Hodur (2004) estimated $130 million in losses due to a leafy spurge infestation in the Dakotas, Montana, and Wyoming. However, the contribution of N deposition to these losses is still uncertain. To address this uncertainty, Finnoff, Strong, and Tschirhart (2008) developed a bioeconomic simulation model that captures the interrelationships between N deposition, invasive grasses (leafy spurge and cheatgrass), and cattle grazing practices. The model specifically examines how N deposition, in conjunction with cattle stocking practices, contributes to these types of infestations and the resulting economic losses. The main conclusions from the model are qualitative. They find that the effect of N deposition on the spread of invasive grass depends importantly on stocking practices, such that infestations due to a given level of N are less likely to occur with lower stocking rates. These findings imply that N deposition ultimately reduces rangeland productivity, either by lowering the optimal grazing rates or by promoting invasives that reduce rangeland productivity.

5.2.2 Effects on Cultural Services

The primary cultural ecosystem services affected by terrestrial enrichment in CSS, MCF, and western grassland ecosystems are expected to be recreation, aesthetic, and nonuse values.

5.2.2.1 Recreation

National parks, forests, grasslands, and monuments are a major source of recreational services in the western United States, and many of these areas include terrestrial ecosystems potentially threatened by nutrient enrichment. Three national parks and monuments in California contain CSS, including Cabrillo National Monument, Channel Islands National Park, and Santa Monica National Recreation Area. All three parks showcase CSS habitat with educational programs and information provided to visitors, guided hikes, and research projects focused on understanding and preserving CSS. Together a total of 1,456,879 visitors traveled through these three parks in 2008. MCF is highlighted in Sequoia and Kings Canyon National Park, Yosemite National Park, and Lassen Volcanic National Park, where a total of 5,313,754 people visited in 2008.

Grasslands are present in most national park and forest lands in the western United States; however, 20 areas are specifically designated as National Grasslands. These lands, which are mostly located in the Great Plains region, currently consist of almost 4 million acres. They receive over 1 million visitors every year and primarily support activities such as hiking and wildlife viewing (http://www.fs.fed.us/grasslands/).

In addition, numerous state and county parks encompass CSS, MCF, and grassland habitat. Visitors to these parks engage in activities such as camping, hiking, attending educational programs, horseback riding, wildlife viewing, water-based recreation, and fishing.

For example, California's Torrey Pines State Natural Reserve protects CSS habitat, and its Great Valley Grasslands State Park preserves native grasslands of the central valley.

Primarily because of data limitation, it is very difficult to estimate the value of recreations benefits that would result from reductions in terrestrial enrichment. One reason is that recreation statistics are not specifically collected or reported for vulnerable areas such as CSS, MCF, or western grasslands. Nevertheless, data for the entire state of California, which includes all three ecosystems, can provide a point of reference for the potential magnitude of these recreation benefits.

Table 5-1 reports selected land-based recreation statistics for the entire state of California in 2006 (DOI, 2007). There were over 3.3 million hunting days and 45 million wildlife viewing days away from home. Using Kaval and Loomis (2003) day-value estimates for the Pacific Coast region of the United States,—$50.10 and $79.81 from 18 and 23 studies, respectively—the total benefits in 2006 from hunting and wildlife viewing away from home in California were approximately $169 million and $3.6 billion, respectively. In addition, data from California State Parks (2003) indicate that in 2002, 68.7% of adult residents participated in trail hiking for an average of 24.1 days per year. Applying these same rates to Census estimates of the California adult population in 2007 suggests that there were roughly 453 million days of hiking by residents in California in 2007. According to Kaval and Loomis (2003), the average value of a hiking day in the Pacific Coast region is $25.59, based on a sample of 49 studies. Multiplying this average day value by the total participation estimate indicates that the aggregate annual benefit for California residents from trail hiking in 2007 was $11.59 billion.

5.2.2.2 Aesthetic

Beyond the recreational value, CSS, MCF, and western grassland landscapes provide aesthetic services to nearby residents. Aesthetic services not related to recreation include the view of the landscape from houses, as individuals commute, and as individuals go about their daily routine in a nearby community. Studies find that scenic landscapes are capitalized into the price of housing.

Although we know of no studies that look at the value of housing as a function of the view in landscapes that include CSS or MCF, other studies document the existence of housing price premia associated with proximity to forest and open space (Acharya and Bennett, 2001; Geoghegan, Wainger, and Bockstael, 1997; Irwin, 2002; Mansfield, et al., 2005; Smith, Poulos, and Kim, 2002; Tyrvainen and Miettinen, 2000). Aesthetic losses (benefits) associated with the presence (absence) of invasive species should also be capitalized into home values (Horsch and

Table 5-1. Land-Based Recreational Activities in California in 2006 by Residents and Nonresidents

Activities in California by Residents and Nonresidents	
Hunting	
Hunters	**281,000**
Days of hunting	3,376,000
Average days per hunter	12
Wildlife Watching	
Total wildlife-watching participants	**6,270,000**
Away-from-home participants	2,894,000
Around-the-home participants	5,259,000
Days of participation away from home	45,010,000
Average days of participation away from home	16
Activities in California by Residents	
Hunting	
Hunters	**274,000**
Days of hunting	3,339,000
Average days per hunter	12
Wildlife Watching	
Total wildlife-watching participants	**5,704,000**
Away-from-home participants	2,328,000
Around-the-home participants	5,259,000
Days of participation away from home	41,436,000
Average days of participation away from home	18

Source: U.S. Department of the Interior, Fish and Wildlife Service, and U.S. Department of Commerce, U.S. Census Bureau. 2007. *2006 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation*.

Lewis, 2009); however, to our knowledge, no studies have yet examined these effects in western grassland or CSS areas.

5.2.2.3 Nonuse Value

Nonuse value, also called existence value or preservation value, encompasses a variety of motivations that lead individuals to place value on environmental goods or services that they do not use. The values individuals place on protecting rare species, rare habitats, or landscape types that they do not see or visit and that do not contribute to the pleasure they get from other activities are examples of nonuse values. Although measuring the public's WTP to protect endangered species poses theoretical and technical challenges, it is clear that the public places a value on preserving endangered species and their habitat. Data on charitable donations, survey results, and the time and effort different individuals or organizations devote to protecting species and habitat suggest that endangered species have intrinsic value to people beyond the value derived from using the resource (recreational viewing or aesthetic value).

CSS, MCF, and western grasslands are home to a number of important and rare species and habitat types; therefore, protecting these ecosystems through reductions in N deposition is expected to provide nonuse benefits. For example, CSS displays richness in biodiversity with more than 550 herbaceous annual and perennial species. Of these herbs, nearly half are endangered, sensitive, or of special status (Burger et al., 2003). Additionally, avian, arthropod, herpetofauna, and mammalian species live in CSS habitat or use the habitat for breeding or foraging.

Figure 5-4 shows communities of CSS and three important federally endangered species. MCF is home to one federally endangered species and a number of state-level sensitive species. Figure 5-5 provides a map of MCF habitat and two threatened and endangered species. The Audubon Society lists 28 important bird areas in CSS habitat and at least 5 in MCF in California (http://ca.audubon.org/iba/index.shtml).[31] The decline in western grasslands in the United States has also contributed to species endangerment, including for the Attwater prairie chicken and the black-capped vireo in Texas (Conner et al., 2001).

To our knowledge, only one study has specifically estimated values for protecting CSS habitat in California. Stanley (2005) uses a CV survey to measure WTP to support recovery plans for endangered species in Southern California. The survey of Orange County, California, residents asked respondents to value the recovery of a single species (the Riverdale fairy shrimp) and a larger bundle of 32 species found in the county. The acquisition of critical habitat and implementation of the recovery plan were the specific goods being valued in the WTP question and the programs would be financed by an annual tax payment. The average WTP for fairy shrimp recovery was roughly $29 (in 2007 dollars) and for all 32 species was $61 per household, depending on the model used. Aggregating benefits (multiplying average household WTP by the number of households in the county) results in total estimated WTP of over $27 million annually for protecting fairy shrimp and $57 million annually for all 32 species.

[31] Important Bird Areas are sites that provide essential habitat for one or more species of bird.

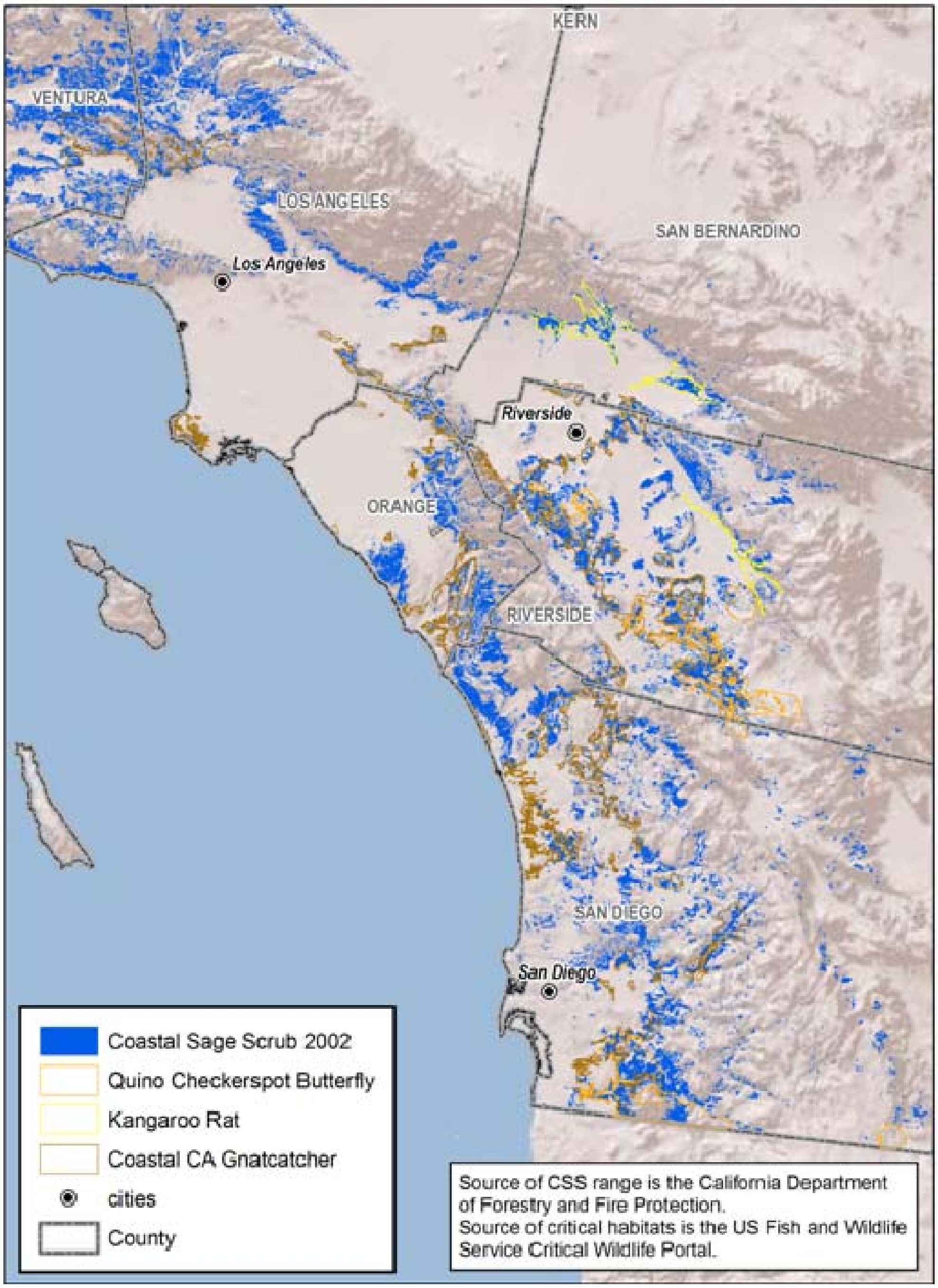

Figure 5-4. Presence of Three Threatened and Endangered Species in California's CSS Ecosystem

Figure 5-5. Presence of Two Threatened and Endangered Species in CA's Mixed Conifer Forest

In a more general study valuing endangered species protection, Loomis and White (1996) synthesize key results from 20 threatened and endangered species valuation studies using meta-analysis methods. They find that annual WTP estimates range from a low of $11 for the striped shiner fish to a high of $178 for the northern spotted owl (in 2007 dollars). None of the studies summarized by Loomis and White are directly associated with CSS, MCF, or western grassland habitats, but the study provides another indication of the value that the public places on preserving endangered species in general.

5.2.3 *Effects on Regulating Services*

Terrestrial ecosystems threatened by N enrichment, including CSS, MCF, and western grasslands, provide a variety of regulating services including protection from soil erosion and climate and water regulation. One of the main services that is potentially threatened by excess N is fire regulation.

As described above, excessive N deposition upsets the balance between native and nonnative plants and grasses. The changing composition of species can result in changes in fire frequency and intensity, as nonnative grasses fuel more frequent and more intense wildfires. More frequent and intense fires also reduce the ability of CSS and native grasses to regenerate after a fire and increase the proportion of nonnative grasses (EPA, 2008). In MCF ecosystems. Excess N deposition leads to changes in the forest structure, such as increased density and loss of root biomass, which in turn can result in more intense fires (EPA, 2008).

Nutrient enrichment is, therefore, likely to be one of the contributing factors to wildfires, which represent a serious threat in the western United States and cause billions of dollars in damage. For example, over the 5-year period from 2004 to 2008, Southern California experienced, on average, over 4,000 fires a year burning, on average, over 400,000 acres (National Association of State Foresters [NASF], 2009). Improved fire regulation leads to short-term and long-term benefits. The short-term benefits include the value of avoided residential property damages; avoided damages to timber, rangeland, and wildlife resources; avoided losses from fire-related air quality impairments; avoided deaths and injury due to fire; improved outdoor recreation opportunities; and savings in costs associated with fighting the fires and protecting lives and property. For example, the California Department of Forestry and Fire Protection (CAL FIRE) estimated that average annual losses to homes due to wildfire from 1984 to 1994 were $163 million per year (CAL FIRE, 1996) and were over $250 million in 2007 (CAL FIRE, 2008). In fiscal year 2008, CAL FIRE's costs for fire suppression activities were nearly $300 million (CAL FIRE, 2008). Therefore, even a 1% reduction in these damages and costs would imply benefits of over $5 million per year.

Figure 5-6 is a map of the overlap between fire threat and CSS habitat. CSS overlaps with areas of very to extremely high fire threat. As shown in Figure 5-7, MCF is also found in some areas closer to the coast with extremely high fire threat and in areas up in the mountains also under very high fire threat.

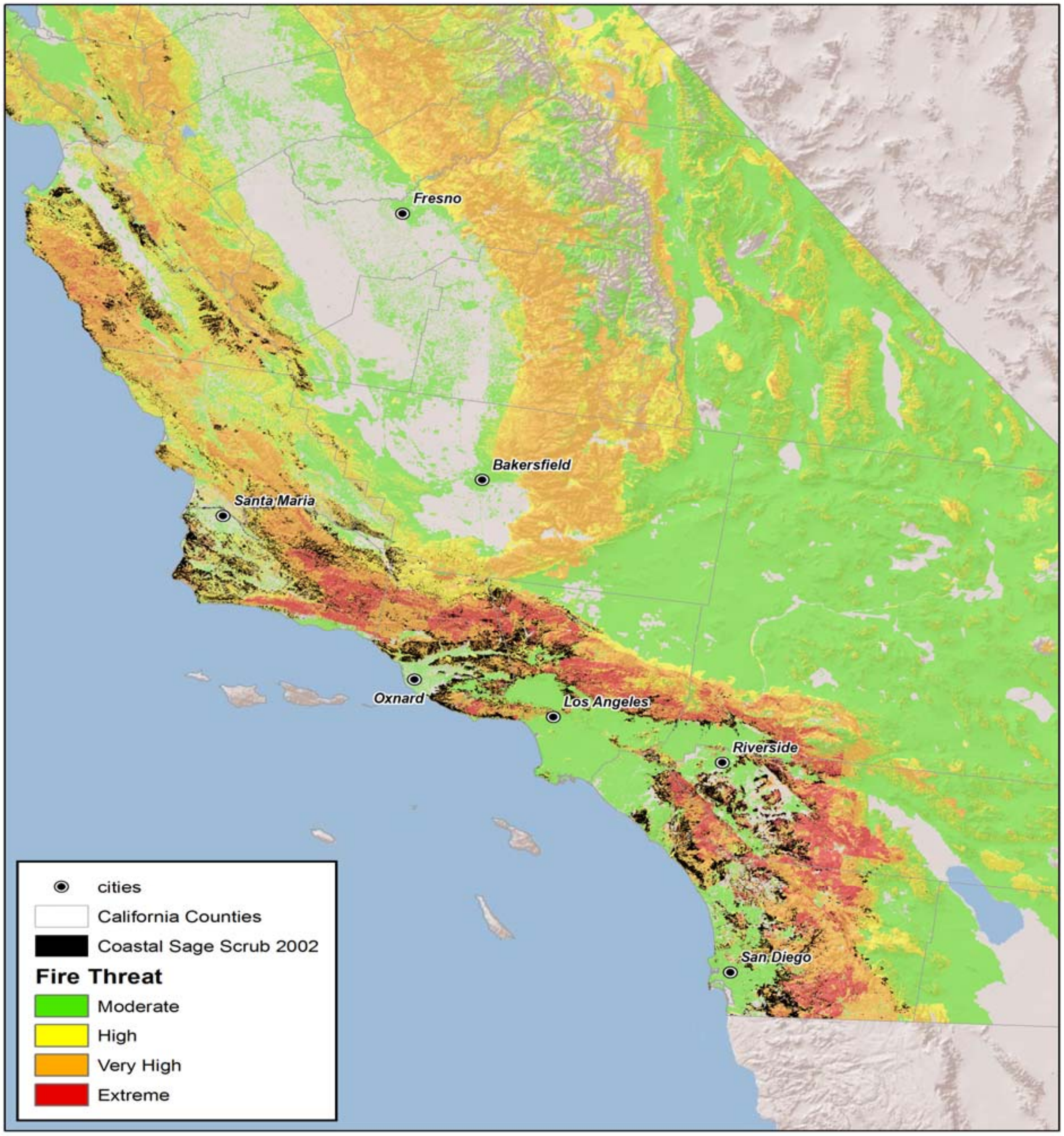

Figure 5-6. CSS Areas and Fire Threat

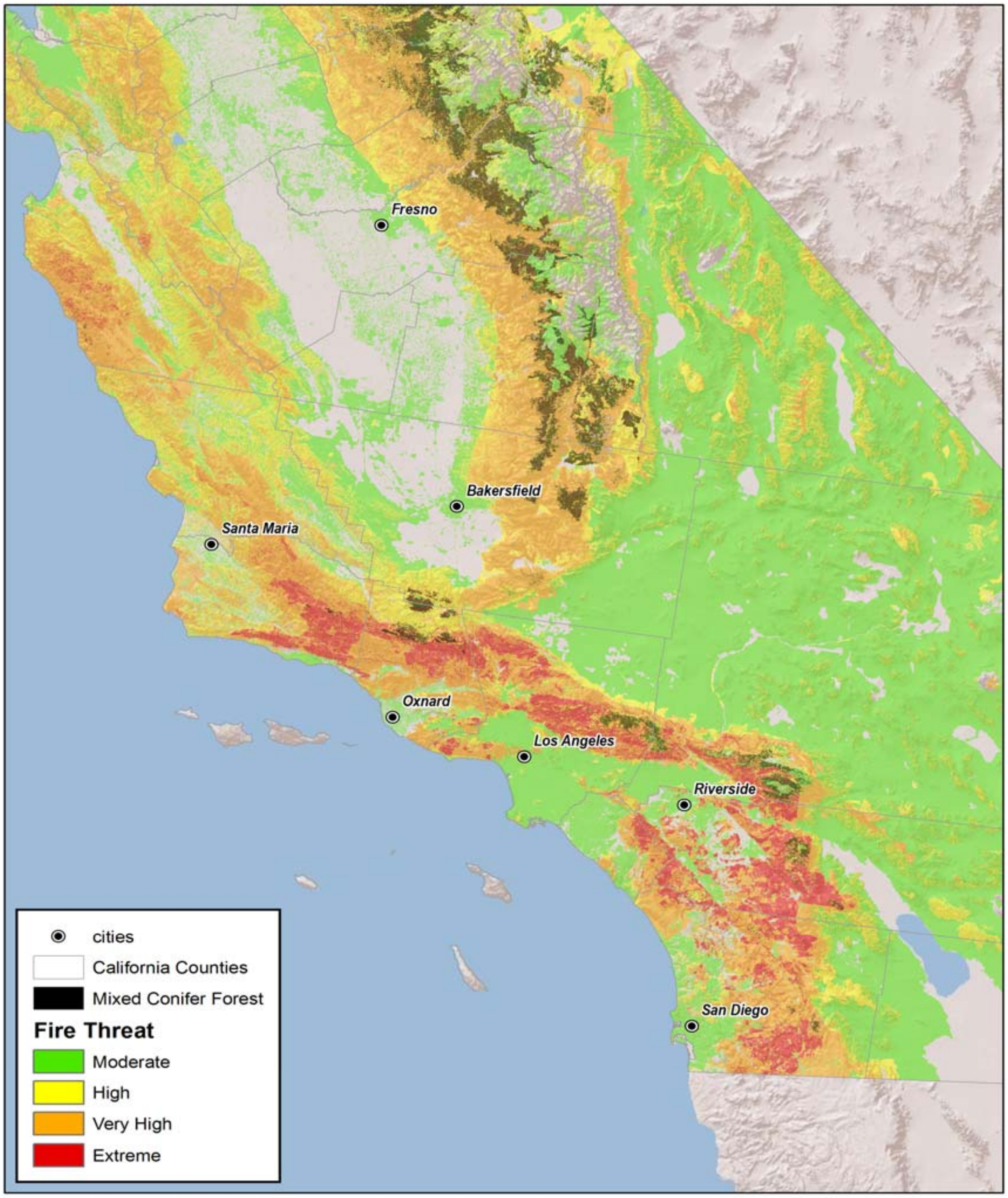

Figure 5-7. MCF Areas and Fire Threat

In western grasslands, invasions of cheat grass, which may be facilitated by the effects of excess N deposition, have become a serious fire threat. According to Knapp (1996), the costs of controlling fires associated with cheat grass were $20 million per year.

In the long term, decreased frequency of fires could result in an increase in property values in fire-prone areas. Mueller, Loomis, and González-Cabán (2007) conducted a hedonic pricing study to determine whether increasing numbers of wildfires affect house prices in southern California. They estimated that house prices would decrease 9.71% ($30,693 in 2007 dollars) after one fire and 22.7% ($71,722; $102,417 cumulative) after a second wildfire within 1.75 miles of a house in their study area. After the second fire, the housing prices took between 5 and 7 years to recover. The results come from a sample of 2,520 single-family homes located within 1.75 miles of one of five fires during the 1990s.

Long-term decreases in wildfire risks are also expected to provide outdoor recreation benefits. The empirical literature contains several articles measuring the relationship between wildfires and recreation values; however, very few address fires in California, particularly in CSS areas. One exception is Loomis et al. (2002), which estimates the changes in deer harvest and deer hunting benefits resulting from controlled burns or prescribed fire in the San Bernardino National Forest in Southern California. Using a CV survey of deer hunters in California, they estimated that the net economic value of an additional deer harvested is, on average, $122 (in 2007 dollars). Based on predicted changes in deer harvest in response to a prescribed fire, they estimated that annual economic benefits for an additional 1,000 acres of prescribed burning ranges from $3,328 to $3,893.

5.3 References

Acharya, G., and L.L. Bennett. 2001. "Valuing Open Space and Land-Use Patterns in Urban Watersheds." *Journal of Real Estate Finance and Economics* 22(2/3):221-237.

Allen, E.B., P.E. Padgett, A. Bytnerowicz, and R. Minnich. 1998. *Nitrogen Deposition Effects on Coastal Sage Vegetation of Southern California*. General Tech. Rep. PSW-GTR-166. U.S. Department of Agriculture, Forest Service, Albany, CA.

Allen, E.B., L. E. Rao, R. J. Steers, A. Bytnerowicz, and M. E. Fenn. 2009. Impacts of atmospheric nitrogen deposition on vegetation and soils in Joshua Tree National Park. In R.H. Webb, L.F. Fenstermaker, J.S. Heaton, D.L. Hughson, E.V. McDonald, and D.M. Miller, eds. The Mojave Desert: Ecosystem Processes and Sustainability. University of Nevada Press, Las Vegas.

Allen, E.B., P.J. Temple, A. Bytnerowicz, M.J. Arbaugh, A.G. Sirulnik, and L.E. Rao. 2007. Patterns of understory diversity in mixed coniferous forests of Southern California impacted by air pollution. *TheScientificWorldJournal 7*(S1):247-263.

Burger, J.C., R.A. Redak, E.B. Allen, J.T. Rotenberry, and M.F. Allen. 2003. "Restoring Arthropod Communities in Coastal Sage Scrub." *Conservation Biology* 17(2):460-467.

CAL FIRE (California Department of Forestry and Fire Protection). 1996. California Fire Plan. Available at http://cdfdata.fire.ca.gov/fire_er/fpp_planning_cafireplan.

CAL FIRE (California Department of Forestry and Fire Protection). 2008. CAL FIRE 2007 Wildland Fire Summary. California Department of Fish and Game. n.d. Available at http://www.dfg.ca.gov/habcon/nccp/images/region.gif.

California State Parks. 2003. *Public Opinions and Attitudes on Outdoor Recreation in California 2002: An Element of the California Outdoor Recration Plan.* Sacramento, California.

Clark, C.M., and D. Tilman. 2008. Loss of plant species after chronic low-level nitrogen deposition to prairie grasslands. *Nature 451*:712–715.

Conner, R., A. Seidl, L. VanTassel, N. Wilkins. 2001. “United States Grasslands and Related Resources: An Economic and Biological Trends Assessment.”

Burger, J.C., R.A. Redak, E.B. Allen, J.T. Rotenberry, and M.F. Allen. 2003. “Restoring Arthropod Communities in Coastal Sage Scrub.” *Conservation Biology* 17(2):460-467.

CAL FIRE (California Department of Forestry and Fire Protection). 1996. California Fire Plan. Available at http://cdfdata.fire.ca.gov/fire_er/fpp_planning_cafireplan.

CAL FIRE (California Department of Forestry and Fire Protection). 2008. CAL FIRE 2007 Wildland Fire Summary. California Department of Fish and Game. n.d. Available at http://www.dfg.ca.gov/habcon/nccp/images/region.gif.

Fenn, M.E., M.A. Poth, S.L. Schilling, and D.B. Grainger. 2000. Throughfall and fog deposition of nitrogen and sulfur at an N-limited and N-saturated site in the San Bernardino Mountains, southern California. *Canadian Journal of Forest Research 30*:1476–1488.

Fenn, M.E., S. Jovan, F. Yuan, L. Geiser, T. Meixner, and B.S. Gimeno. 2008. Empirical and simulated critical loads for nitrogen deposition in California mixed conifer forests. *Environmental Pollution 155*(3):492–511.

Finnoff, D., A. Strong, and J. Tschirhart. 2008. A bioeconomic model of cattle stocking on rangeland threatened by invasive plants and nitrogen deposition. *Amer. J. Agr. Econ.* 90(4): 1074–1090.

Geoghegan, J., L.A. Wainger, and N.E. Bockstael. 1997. “Spatial Landscape Indices in a Hedonic Framework: An Ecological Economics Analysis Using GIS.” *Ecological Economics* 23:251-264.

Grace, J.B., M.D. Smith, S.L. Grace, S.L. Collins, and T.J. Stohlgren. 2001. Interactions between fire and invasive plants in temperate grasslands of North America. P. 40-65 in K.E.M. Galley and T.P. Wilson, eds. Proceedings of the Invasive Species Workshop: the Role of Fire in the Control and Spread of Invasive Species. Fire Conference 2000: the First National Congress on Fire Ecology, Prevention, and Management. Miscellaneous Publication # 11, Tall Timbers Research Station, Tallahassee, FL.

Grulke, N.E., C.P. Andersen, M.E. Fenn, and P.R. Miller. 1998. Ozone exposure and nitrogen deposition lowers root biomass of ponderosa pine in the San Bernardino Mountains, California. *Environmental Pollution 103*:63–73.

Grulke N.E., C. Andersen, W.E. Hogsett. 2001. Seasonal changes in above- and belowground carbohydrate concentrations of ponderosa pine along a pollution gradient. *Tree Physiology 21*:175–184.

Horsch, E. J, and D. J Lewis. 2009. The Effects of Aquatic Invasive Species on Property Values: Evidence from a Quasi-Experiment. *Land Economics* 85, no. 3: 391.

Irwin, E.G. 2002. "The Effects of Open Space on Residential Property Values." *Land Economics* 78(4):465-480.

Kaval, P., and J. Loomis. 2003. *Updated Outdoor Recreation Use Values with Emphasis on National Park Recreation*. Final Report October 2003, under Cooperative Agreement CA 1200-99-009, Project number IMDE-02-0070.

Keeler-Wolf, T. 1995. Post-fire emergency seeding and conservation in Southern California shrublands. Pp. 127–139 in *Brushfires in California Wildlands: Ecology and Resource Management*. Edited by J.E. Keeley and T. Scott. International Association of Wildland Fire, Fairfield, WA.

Knapp, Paul A. 1996. Cheatgrass (Bromus tectorum L) dominance in the Great Basin Desert : History, persistence, and influences to human activities. *Global Environmental Change* 6, no. 1 (April): 37-52.

Leistritz, F. L., D. A. Bangsund, and N. M. Hodur. 2004. "Assessing the Economic Impact of Invasive Weeds: The Case of Leafy Spurge (*Euphorbia esula*)." *Weed Technology* 18:1392-1395.

Leitch, J. A, F. L Leistritz, and D. A Bangsund. 1996. Economic effect of leafy spurge in the upper Great Plains: methods, models, and results. *Impact Assessment* 14: 419–434.

Loomis, J., D. Griffin, E. Wu, and A. González-Cabán. 2002. "Estimating the Economic Value of Big Game Habitat Production from Prescribed Fire Using a Time Series Approach." *Journal of Forest Economics* 2:119-29.

Loomis, J.B., and D.S. White. 1996. "Economic Benefits of Rare and Endangered Species: Summary and Meta-Analysis." *Ecological Economics* 18(3):197-206.

Mansfield, C.A., S.K. Pattanayak, W. McDow, R. MacDonald, and P. Halpin. 2005. "Shades of Green: Measuring the Value of Urban Forests in the Housing Market." *Journal of Forest Economics* 11(3):177-199.

Minnich, R.A., M.G. Barbour, J.H. Burk, and R.F. Fernav. 1995. Sixty years of change in conifer forests of the San Bernardino Mountains: Reconstruction of California mixed conifer forests prior to fire suppression. *Conservation Biology 9*:902–914.

Minnich, R.A. 2007. Southern California coniferous forest. Chapter 18 in *Terrestrial Vegetation of California, 3rd Edition*. Edited M.G. Barbour, T. Keeler-Wolf, and A.S. Schoenherr. University of California Press, Berkeley, CA.

Monaco, T. A., D.A. Johnson, J.M. Norton, T.A. Jones, K.J. Connors, J.B. Norton, and M.B. Redinbaugh. 2003. Contrasting responses of Intermountain West grasses to soil nitrogen. J. Range Manage. 56: 282-290.

Mueller, J., J. Loomis, and A. González-Cabán. 2007. "Do Repeated Wildfires Change Homebuyers' Demand for Homes in High-Risk Areas? A Hedonic Analysis of the Short and Long-Term Effects of Repeated Wildfires on House Prices in Southern California." *Journal of Real Estate Finance and Economics*, 1-18.

National Association of State Foresters (NASF). 2009. Quadrennial Fire Review 2009.Washington, DC: NASF. Quadrennial Fire and Fuel Review Final Report 2009. National Wildfire Coordinating Group Executive Board.

Padgett, P.E., and E.B. Allen. 1999. Differential responses to nitrogen fertilization in native shrubs and exotic annuals common to Mediterranean coastal sage scrub of California. *Plant Ecology 144*:93–101.

Rao, L.E., D.R. Parker, A. Bytnerowicz, E.B. Allen. 2009. Nitrogen mineralization across an atmospheric nitrogen deposition gradient in Southern California deserts. Journal of Arid Environments 73: 920–930.

Schwinning, S,.B.I. Starr, N.J. Wojcik, M.E. Miller, J.E. Ehleringer, and R.L. Sanford Jr. 2005. Effects of Nitrogen Deposition on an Arid Grassland in the Colorado Plateau Cold Desert. Rangeland Ecol Manage 58:565–574.

Smith, V.K., C. Poulos, and H. Kim. 2002. "Treating Open Space as an Urban Amenity." *Resource and Energy Economics* 24:107-129.

Stanley, D.L. 2005. "Local Perception of Public Goods: Recent Assessments of Willingness-to-Pay for Endangered Species." *Contemporary Economic Policy* 23:165-179.

Stevens, C.J., L.C. Maskell, S.M. Smart, S.J.M. Caporn, N.B. Dise, D.J.G. Gowing. 2009. Identifying indicators of atmospheric nitrogen deposition impacts in acid grasslands Biological Conservation 142; 2069–2075.

Suding, K.N., S.L. Collins, L. Gough, C. Clark, E.E. Cleland, K.L. Gross, D.G. Milchunas, and S.Pennings. 2005. Functional- and abundance-based mechanisms explain diversity loss due to N fertilization. PNAS 102 (12): 4387-4392.

Takemoto, B.K., A. Bytnerowicz, and M.E. Fenn. 2001. Current and future effects of ozone and atmospheric nitrogen deposition on California's mixed conifer forests. *Forest Ecology and Management 144*:159–173.

Tyrvainen, L., and A. Miettinen. 2000. "Property Prices and Urban Forest Amenities." *Journal of Economics and Environmental Management* 39:205-223.

U.S. Department of the Interior, Fish and Wildlife Service, and U.S. Department of Commerce, U.S. Census Bureau. 2007. *2006 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation*.

U.S. Department of the Interior, Fish and Wildlife Service, and U.S. Department of Commerce, U.S. Census Bureau. 2007. *2006 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation*.

U.S. Environmental Protection Agency (EPA). 2008. Integrated Science Assessment for Oxides of Nitrogen and Sulfur–Environmental Criteria.EPA/600/R-08/082. U.S. Environmental Protection Agency, Office of Research and Development, National Center for Environmental Assessment, RTP Division, Research Triangle Park, NC.

Weiss, S.B. 1999. Cars, cows, and checkerspot butterflies: nitrogen deposition and management of nutrient-poor grasslands for a threatened species. Conservation Biology 13(6):1476–1486.

Westman, W.E. 1981a. Diversity relations and succession in Californian coastal sage scrub. *Ecology 62*:170–184.

Westman, W.E. 1981b. Factors influencing the distribution of species of Californian coastal sage scrub. *Ecology 62*:439–455.

Wood, Y., T. Meixner, P.J. Shouse, and E.B. Allen. 2006. Altered Ecohydrologic response drives native shrub loss under conditions of elevated N-deposition. *Journal of Environmental Quality 35*:76–92.

SECTION 6
MERCURY METHYLATION

6.1 Overview of Affected Ecosystem Services

In addition to acidification and nutrient enrichment effects, S deposition is a source of concern for ecosystem health through its contribution to the mercury methylation process. This is the process that converts elemental (inorganic) mercury (Hg^{+2}) into its more potent and toxic form—methylmercury ($MeHg^{+}$). The diagram in Figure 6-1 is a conceptual model that illustrates the links between S deposition, mercury-related damages to ecosystems, and subsequent impairment of ecosystem services.

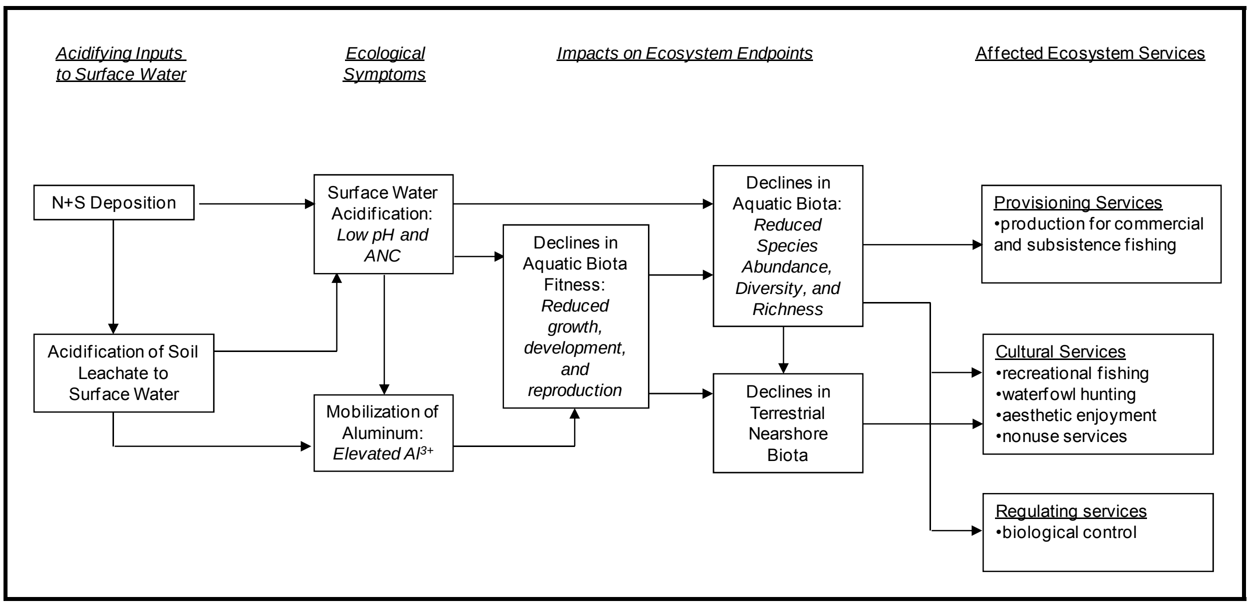

Figure 6-1. Conceptual Diagram of Ecosystem Service Impairments Associated with Mercury Methylation[a]

[a] Bold arrows represent the stronger and better established cause-and-effect relationships.

Mercury methylation occurs in aquatic systems related to all major waterbody types, rivers, lakes, estuaries, and wetlands. Most detailed site-specific case studies have focused on lake ecosystems (the lake waterbodies and small catchment areas for the lakes), where methylation can be more readily analyzed in anoxic waters and sediments.

The mercury methylation process is represented in Figure 6-2. When S compounds (in particular SO_4^{2}) enter lakes through direct deposition or S-containing runoff, they can trigger a

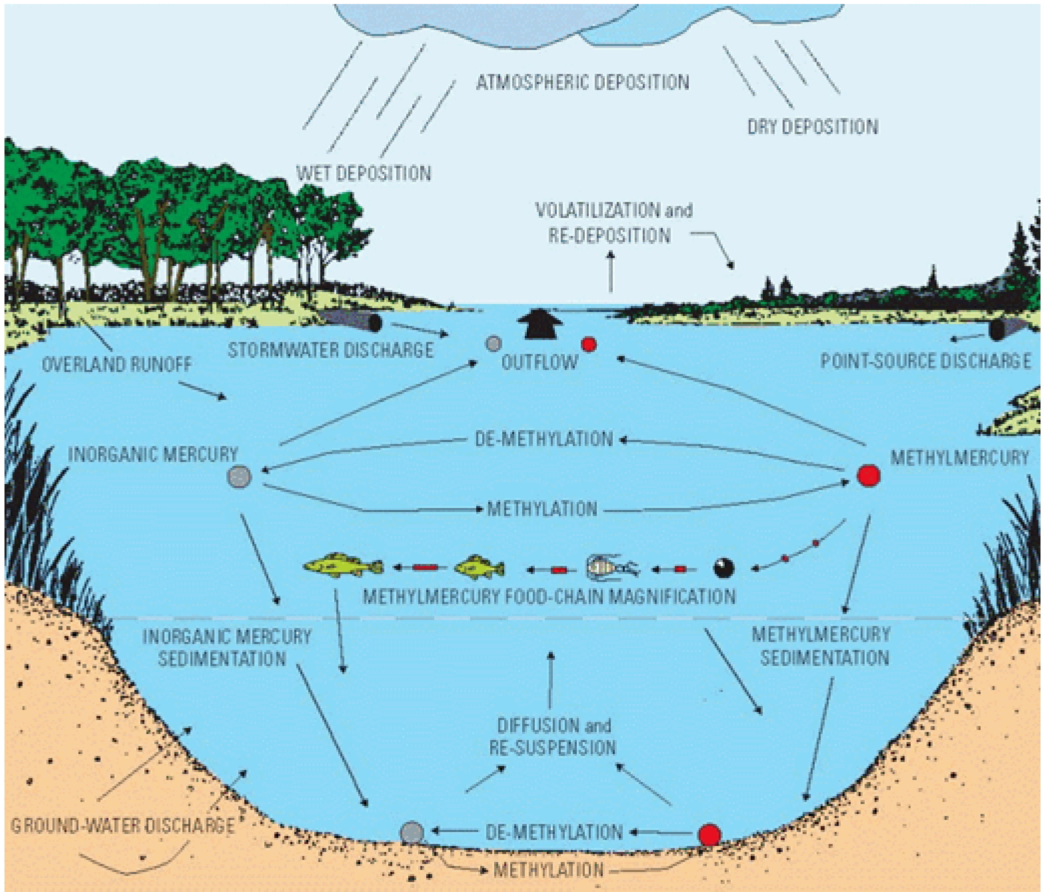

Figure 6-2. The Mercury Cycle in an Ecosystem

Source: U.S. Geological Survey (USGS). 2006. *Investigations and Monitoring of Mercury in Indiana by the U.S. Geological Survey*. Indianapolis, IN: U.S. Department of the Interior, U.S. Geological Survey, Indiana Water Science Center. http://in.water.usgs.gov/mercury.

methylation process if S-reducing bacteria (SRB) are also present in sufficient quantities. Under these conditions, when inorganic mercury is also present, the SRB transform the inorganic mercury to $MeHg^+$. The rate of transformation and resulting magnitude of $MeHg^+$ accumulation in sediments and water depends on several biogeochemical factors, such as temperature, presence of organic matter, and waterbody limnology. The $MeHg^+$ present in the water column and sediments is then ingested by aquatic organisms and moves up the food chain where it accumulates in the tissue of fish, birds, and mammals. Unlike other toxic substances that can enter organisms in local food chains through direct bioconcentration, the buildup of mercury in higher organisms involves almost exclusively the $MeHg^+$ form of mercury and the bioaccumulation of the toxicant through trophic food webs.

The resulting $MeHg^+$ exposures can cause a variety of toxic effects in fish and wildlife, including reproductive and neurological disorders. There are also clearly potential impacts for human population groups, which can detract from the recreational amenities of lakes and other aquatic resources and can also lead to human health risks.

Although the existing science provides strong evidence of a causal link between S deposition and rates of $MeHg^+$ accumulation in lake ecosystems, models are not currently developed to quantify this relationship adequately without considerable uncertainty on a site-specific basis to (1) measure the contribution of past and current S loads to the existing spatial distribution of mercury concentrations in surface water and fish tissues or (2) predict changes in these concentration as a result of future reductions in S deposition.

Most operational models for mercury bioaccumulation analysis do not attempt to model the actual methylation processes in any detail. Input parameters related to the methylation process must be provided, and considerable uncertainly can be introduced into the models where detailed site-specific information is not readily available. The way $MeHg^+$ bioaccumulates in aquatic food chains also requires specifying the structure of local food webs from algae and plankton to forage fishes and then to sports fishes. For a complete process-modeling framework capturing the details of the bacteria-mediated methylation process, a model would need capabilities to factor in site-specific S and mercury deposition data. Where these types of site-specific data are lacking, additional uncertainty would be added to the model outputs.

EPA is in the process of expanding the functionality of its modeling frameworks designed to handle mercury bioaccumulation. The main objective is to implement modeling frameworks that can handle aquatic systems for local watershed units on the spatial scale of entire HUC8 subbasins (EPA, 2009). These new modeling frameworks include the Watershed Characterization System Mercury Loading Model (WCS-MLM) and the Grid-Based Mercury Loading Model (GBMM and the Watershed Scale WCS-GBMM). Both of these new frameworks will be better equipped to simulate mercury fate and transport processes in rivers as well as in smaller lake catchments. The GBMM uses a grid (raster) approach that will provide better interfacing with the prediction grids of CMAQ and also enhance modeling processes involving wetlands and soils. At the present time, however, the Spreadsheet-based Ecological Risk Assessment for the Fate of Mercury (SERAFM) reflects the best operational models. The series of submodules within SERAFM include mercury loading (watershed and atmospheric deposition), abiotic and biotic solids balance (soil erosion, settling, burial, and resuspension), equilibrium partitioning, water body mercury processes, and wildlife risk calculations. SERAFM (http://www.epa.gov/ceampubl/swater/serafm/index.html) has been applied in a number of regulatory analyses (EPA, 2005), and where suitable site-specific case study data are available, it is well suited to simulating processes for small lakes and their immediate catchment areas (Knightes, 2007; Knightes and Ambrose, 2006).

Given the limitations of currently available modeling capabilities, it is still possible to describe in more qualitative terms the expected effects of mercury methylation on specific ecosystem services in the United States using geospatial analysis techniques that take advantage of the mercury sensitivity ranking indicators (Myers et al., 2007) and leverage relevant analyses from previous regulatory initiatives (EPA, 2005). One important analytical capability is to identify watershed areas with appreciable vulnerability to mercury methylation, where additional inputs of S from atmospheric deposition would be of concern for their potential role in increasing the bioaccumulation of mercury in food chains. EPA has worked with the U.S. Geological Survey (USGS) to provide mercury methylation vulnerability rankings for most of the subbasin (HUC8) watershed units in the conterminous United States (Myers et al., 2007). HUC8s are assigned indicator scores based on analyses of the sulfate concentrations in ambient water, acid neutralizing capacity, DOC, pH, mercury species concentration, and soil types. These scores allow one to define "methylation potential" as a relative indicator of the likelihood that mercury deposited in a given HUC8 subbasin aquatic ecosystem will be converted to $MeHg^+$ and bioaccumulate in fish.

The map in Figure 6-3 shows mercury sensitivity rankings (high, medium, and low) for aquatic systems related to HUC8 subbasins. In this presentation, the quintile classifications in maps provided in Myers et al. (2007) are converted to a classification with three rankings (high, medium, and low), where the high category reflects the upper quintile from the published USGS maps, the medium category reflects the middle two quintiles, and the low category reflects the lowest two original quintile categories, along with HUC8s for which data are inadequate to develop an indicator score.

Given the limitations of currently available modeling capabilities, it is still possible to describe in more qualitative terms the expected effects of mercury methylation on specific ecosystem services in the United States. These are the services that would be enhanced by reductions in S deposition and would, therefore, be a potentially important source of ecological benefits.

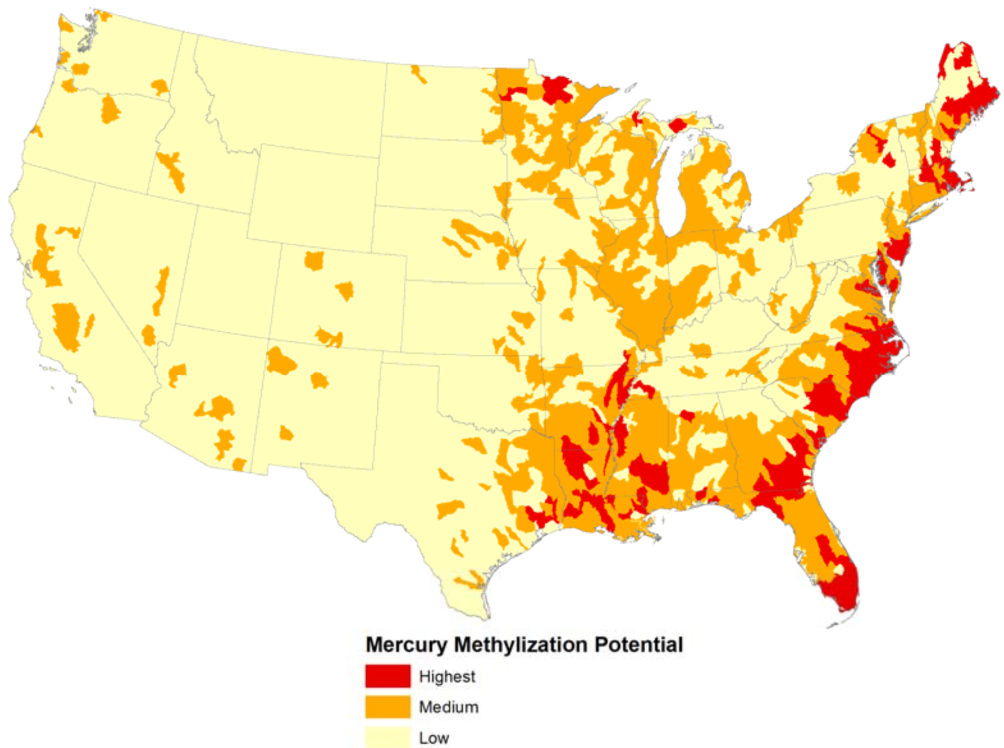

Figure 6-3. Mercury Methylation Potential in U.S. Watersheds

6.1.1 Effects on Provisioning Services

Although lake ecosystems in the United States are not a major source of commercially caught fish, they do provide a source of nutrition for recreational and subsistence fishers and their households. Mercury accumulation in these ecosystems impairs these provisioning services by (1) potentially decreasing the abundance of available fish because of toxic effects on fish health and reproduction and (2) increasing the human health risks (thus reducing the nutritional value) associated with fish consumption.

Accumulation of MeHg in fish tissue is considered a significant threat to the health of both wildlife and humans (Scudder et al., 2009). Microscopic organisms convert inorganic mercury into $MeHg^+$, which accumulates up the food chain in fish, fish-eating animals, and people. Birds and mammals that eat fish are more exposed to $MeHg^+$ than any other animals in water ecosystems. Similarly, predators that eat fish-eating animals are at risk. $MeHg^+$ has been found in eagles, otters, and endangered Florida panthers. Analyses conducted for the Mercury Study Report to Congress (EPA, 1997) suggest that some highly exposed wildlife species are being harmed by $MeHg^+$. Effects of $MeHg^+$ exposure on wildlife can include mortality (death), reduced fertility, and slower growth and development, and abnormal behavior can affect survival, depending on the level of exposure. In addition, research indicates that the endocrine system of fish, which plays an important role in fish development and reproduction, might be altered by the levels of $MeHg^+$ found in the environment.

For humans, it is well established that $MeHg^+$ is a potent neurotoxin at high doses, whereas at low doses, the strongest evidence is for neurodevelopmental effects resulting from prenatal exposures. A number of epidemiological studies have found that exposures before birth, primarily through mothers' fish consumption during pregnancy, are positively associated with cognitive difficulties later in life. The evidence is less clear for postnatal exposures; however, some studies have found increased risks of impaired neurological and sensory functions, and there is mixed evidence regarding cardiovascular, genotoxic, immunotoxic, and carcinogenic effects in adults due to chronic low-level mercury exposures (EPA, 2005).

Nearly all human exposures to $MeHg^+$ in the United States occur through eating fish and shellfish, and a portion of these exposures occur through consuming lake fish. Fish tissue samples drawn from across the country provide widespread evidence of elevated mercury concentrations in freshwater fish. Figure 6-4 shows the spatial patterns for tissue concentrations of $MeHg^+$ averaged for HUC8 subbasins for a base period of 1995 to 2001 (EPA, 2005). Several areas were found to have average concentrations exceeding the human-health criterion of 0.3 micrograms per gram wet weight. In addition, as of 2006 (see Figure 6-5), most states (48; no advisories in Alaska or Wyoming), the District of Columbia, one territory (American Samoa), and two tribes had issued fish-consumption advisories for Hg (EPA, 2007). These advisories represent 14,177,175 lake acres and 882,963 river miles, or 35% of the nation's total lake acreage and about 25% of its river miles.

The health risks associated with consuming mercury-contaminated fish can severely degrade the provisioning services provided by lake ecosystems in at least two ways. First, for those who reduce or avoid lake fish consumption (e.g., as a result of fish consumption advisories), not consuming the fish reduces the nutrition received from these fish (Oken et al., 2003). Second, consuming the fish can damage the health of those who eat the fish or are prenatally exposed.

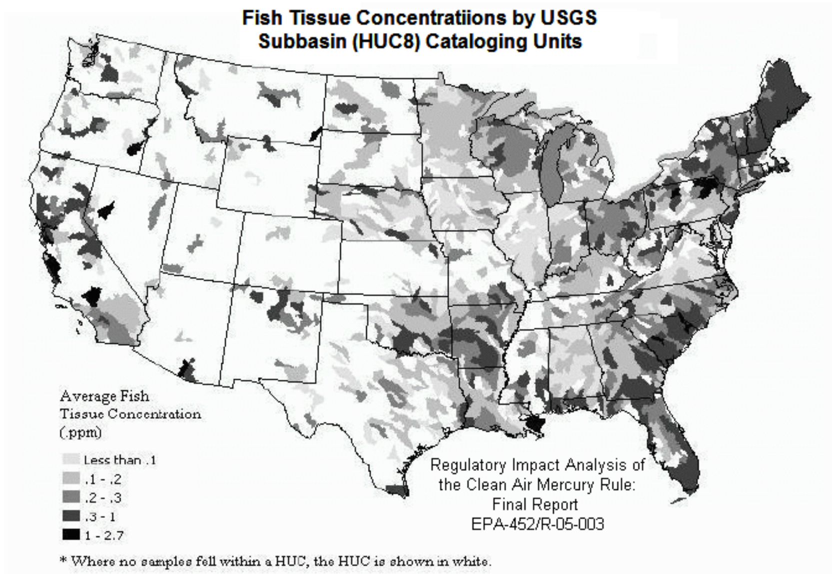

Figure 6-4. Map Showing Average MeHg+ Fish Tissue Concentrations from 1995–2001 for USGS Subbasin (HUC8) Cataloguing Units

Source: U.S. Environmental Protection Agency (EPA). 2005. *Regulatory Impact Analysis of the Final Clean Air Mercury Rule*. EPA-452/R-05-003, March 2005. Research Triangle Park, NC: U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards.

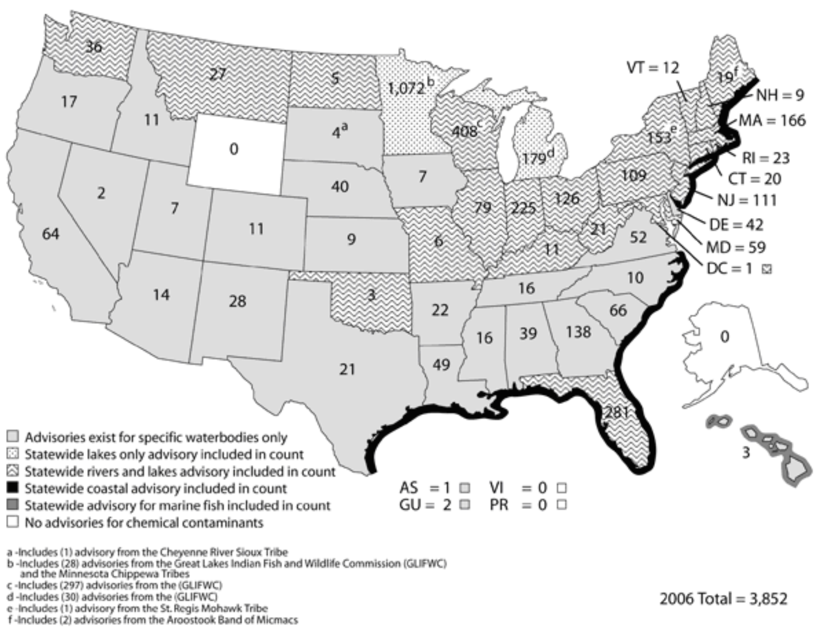

Figure 6-5. Distribution Pattern in 2006 for State Fish Consumption Advisory Listings

Source: U.S. Environmental Protection Agency (EPA). 2007. "National Listing of Fish Advisories Technical Fact Sheet: 2005/06 National Listing Fact Sheet." EPA-823-F-07-003. Washington, DC: U.S. Environmental Protection Agency, Office of Water.

The health risks from mercury exposure are particularly high for groups with relatively high rates of freshwater fish consumption. One such group is recreational anglers who consume their catch. According to FHWAR, over 21 million anglers fished in lakes, reservoirs, and ponds in 2006 for a total of more the 304 million fishing days. Nationwide, anglers kept 50% of the fish they caught (USDOI, 2007).

Subpopulations who engage in subsistence fishing are another potential high-risk group. These subpopulations include many native American groups, and testing has shown that people in these communities who eat significant quantities of fish from local waters with high rates of bioaccumulation of $MeHg^+$ in the aquatic food chains will generally have mercury levels well above the safe limit (Roe, 2003; EPA, 2005). Southeast Asians are another ethnic group whose cultures have traditionally had relatively high rates of fish consumption (EPA, 1997).

Geospatial overlays can be applied to further investigate the spatial relationship between potential high fish consumption groups and areas with high mercury methylation potential. Figure 6-6 shows the spatial distribution of tribal census tracts based on information from the 2000 Census (U.S. Census Bureau, 2009) with the tracts assigned low, medium, or high susceptibility rankings spatially correlated with the subbasin rankings from the aquatic ecosystem sensitivity map. Tribal census tracts are small, relatively permanent statistical subdivisions of a federally recognized American Indian reservation and/or off-reservation trust land. The optimum size for a tribal census tract is 2,500 people; it must contain a minimum of 1,000 people (U.S. Census Bureau, 2009; EPA, 2005).

The tribal tracts located in subbasins with moderate to high ranking represent areas with high *potential* benefits associated with reductions in S deposition. For example, these areas include certain tracts in Minnesota and Wisconsin with significant number of Chippewa. The Chippewa are among the five most populous tribes in the United States, numbering over 100,000 in 2000 and are primarily located in parts of Minnesota, Wisconsin, and Michigan. Other areas with overlap between the presence of tribal groups and high mercury methylation potential are in southern Maine and southern Florida.

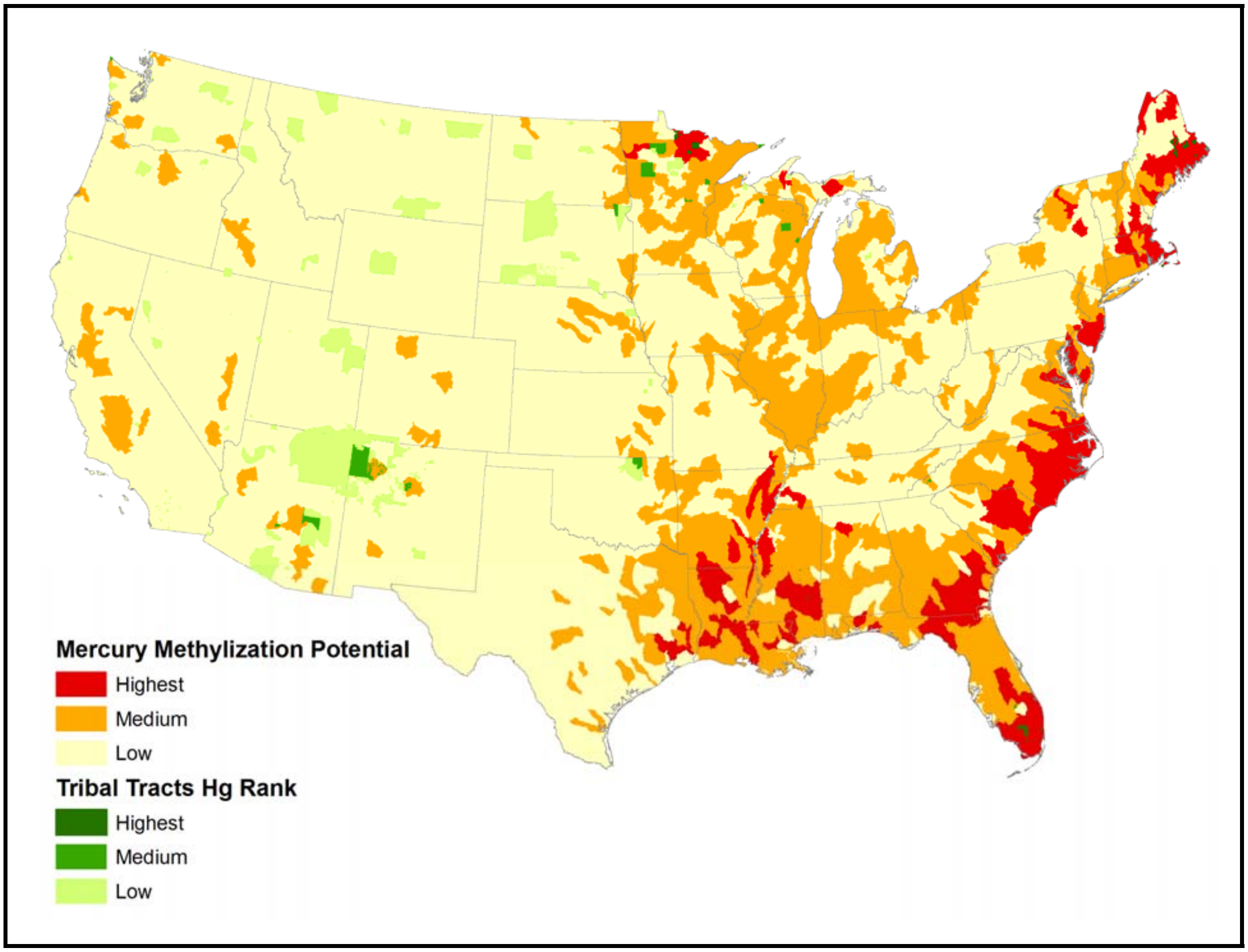

Figure 6-6. Overlay of HUCs with High Mercury Methylation Potential and Tribal Census Tracts

6.1.2 Effects on Cultural Services

In addition to affecting the provisioning services received from lakes and fish consumption, mercury contamination can impair many of the cultural services derived from lakes and the activities associated with these ecosystems. For example, the cultural services and values derived from recreational fishing are adversely affected when anglers change their fishing behavior in response to mercury contamination. Several recreation demand studies have found that fish consumption advisories significantly affect anglers' decisions regarding whether, how often, and where to fish (Jakus, McGuinness, and Krupnick, 2002). As a result, they also significantly reduce the annual benefits derived from recreational fishing for the affected anglers.

Mercury contamination may also have a significant adverse effect on cultural practices and the services derived from these activities. For example, subsistence fishing is closely tied to cultural identity for many native American groups. Freshwater fishing events are also important cultural rituals for certain indigenous group (Roe, 2003). One well-documented case study is in the Midwest, where Chippewa Indians depend heavily on fish for cultural identity, including during ritual ceremonies. Every year the seasonal break up of ice is celebrated through a community-wide feast of walleye fish that are caught during a spearfishing event. Fishes not eaten at the feast are often taken home and frozen for future meals.

6.1.3 Effects on Regulating Services

Inland waters, including lakes, provide a number of regulating services, such as hydrological regime regulation and climate regulation, but mercury methylation is not likely to have a significant degrading effect on many of these specific services. Nevertheless, like the effects of acidification, mercury methylation may affect the biological control services provided by lake ecosystems through their sustenance of delicate aquatic food chains. Through its toxic effects on fish, mercury accumulation impairs these services by disrupting the trophic structure of surface waters and nearshore ecosystems. Although it is difficult to quantify these services, they may be at least partially captured through measures of provisioning and cultural services. For example, these biological control services may serve as “intermediate” inputs that support the production of the “final” provisioning and cultural services described above.

6.2 References

Jakus, P., M. McGuinness, and A. Krupnick. 2002. “The Benefits and Costs of Fish Consumption Advisories for Mercury.” Washington, DC: Resources for the Future.

Knightes, C.D. 2007. “Development and Test Application of a Screening-Level Mercury Fate Model and Tool for Evaluating Wildlife Exposure Risk for Surface Waters with Mercury-Contaminated Sediments (SERAFM).” *Environmental Modelling & Software* 23:495-510.

Knightes, C.D., and R. B. Ambrose, Jr. 2006. “Development of an Ecological Risk Assessment Methodology for Assessing Wildlife Exposure Risk Associated with Mercury-Contaminated Sediments in Lake and River Systems.” Athens, GA: U.S. Environmental Protection Agency, Office of Research and Development, National Exposure Research Laboratory, Ecosystems Research Division. http://www.epa.gov/ceampubl/swater/serafm/SERAFM-dl.htm.

Myers, M.D., M.A. Ayers, J.S. Baron, P.R. Beauchemin, K.T. Gallagher, M.B. Goldhaber, D.R. Hutchinson, J.W. LaBaugh, R.G. Sayre, and S.E. Schwarzbach. 2007. “USGS Goals for the Coming Decade.” *Science* 318:200–201.

Oken, E., K.P. Kleinman, W.E. Berland, S.R. Simon, J.W. Rich-Edwards, and M.W. Gillman. 2003. “Decline in Fish Consumption among Pregnant Women after a National Mercury Advisory.” *Obstetrics and Gynecology* 102(2):346.

Roe, A. 2003. "Fishing for Identity: Mercury Contamination and Fish Consumption among Indigenous Groups in the United States." *Bulletin of Science, Technology & Society* 23(5):368-375.

Scudder, B.C., L.C. Chasar, D.A. Wentz, N.J. Bauch, M.E. Brigham, P.W. Moran, and D.P. Krabbenhoft. 2009. *Mercury in Fish, Bed Sediment, and Water from Streams across the United States, 1998–2005*. U.S. Geological Survey Scientific Investigations Report 2009–5109. http://pubs.usgs.gov/sir/2009/5109/.

U.S. Census Bureau. 2009. 2000 Tribal Census Tracts. Cartographic Boundary files. Washington, DC: Department of Commerce, Census Bureau, Geography Division. http://www.census.gov/geo/www/cob/tr_metadata.html#tct_meta.

U.S. Department of the Interior, Fish and Wildlife Service, and U.S. Department of Commerce, U.S. Census Bureau. 2007. *2006 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation*.

U.S. Environmental Protection Agency (EPA). 1997. *Mercury Study Report to Congress: Vol. 3—Fate and Transport of Mercury in the Environment*. EPA-452/R-97-005. Washington, DC: U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards and Office of Research and Development.

U.S. Environmental Protection Agency (EPA). 2005. *Regulatory Impact Analysis of the Final Clean Air Mercury Rule*. EPA-452/R-05-003, March 2005. Research Triangle Park, NC: U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards.

U.S. Environmental Protection Agency (EPA). 2007. "National Listing of Fish Advisories Technical Fact Sheet: 2005/06 National Listing Fact Sheet." EPA-823-F-07-003. Washington, DC: U.S. Environmental Protection Agency, Office of Water.

U.S. Environmental Protection Agency (EPA). 2009. *Mercury Models for Analysis of Mercury Fate*. Athens, GA: U.S. Environmental Protection Agency, Office of Research and Development, National Exposure Research Laboratory, Ecosystems Research Division.

U.S. Geological Survey (USGS). 2006. *Investigations and Monitoring of Mercury in Indiana by the U.S. Geological Survey*. Indianapolis, IN: U.S. Department of the Interior, U.S. Geological Survey, Indiana Water Science Center. http://in.water.usgs.gov/mercury.