Proposal to Designate an Emission Control Area for Nitrogen Oxides, Sulfur Oxides and Particulate Matter

Technical Support Document

Chapter 3
Impacts of Shipping Emissions on Air Quality, Health and the Environment

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3 Impacts of Shipping Emissions on Air Quality, Health and the Environment

Designation of this Emission Control Area will significantly reduce emissions of SO_X, NO_X and $PM_{2.5}$ and ambient levels of particulate matter and ground-level ozone in large portions of the United States, which will result in substantial benefits to human health and the environment. This chapter describes the pollutants which would be reduced due to the ECA designation and their impacts on human health and ambient air quality as well as the impacts of these pollutants on the environment. Appendix A to Chapter 3 describes the relevant meteorological conditions within the proposed areas that contribute to at-sea emissions being transported to populated areas and contributing to harmful human health and ecological impacts. Appendix B to Chapter 3 presents the expected percent reduction in nitrogen and sulfur deposition in 18 regions of the U.S. due to the proposed ECA.

3.1 Pollutants Reduced by the ECA and their Associated Health Impacts

3.1.1 Description of Pollutants

3.1.1.1 Particulate Matter

Particulate matter (PM) is a generic term for a broad class of chemically and physically diverse substances. It can be principally characterized as discrete particles that exist in the condensed (liquid or solid) phase spanning several orders of magnitude in size. Since 1987, EPA has delineated that subset of inhalable particles small enough to penetrate to the thoracic region (including the tracheobronchial and alveolar regions) of the respiratory tract (referred to as thoracic particles). Current national ambient air quality standards (NAAQS) use $PM_{2.5}$ as the indicator for fine particles (with $PM_{2.5}$ referring to particles with a nominal mean aerodynamic diameter less than or equal to 2.5 μm), and use PM_{10} as the indicator for purposes of regulating the coarse fraction of PM_{10} (referred to as thoracic coarse particles or coarse-fraction particles; generally including particles with a nominal mean aerodynamic diameter greater than 2.5 μm and less than or equal to 10 μm, or $PM_{10-2.5}$). Ultrafine particles are a subset of fine particles, generally less than 100 nanometers (0.1 μm) in aerodynamic diameter.

Particles span many sizes and shapes and consist of hundreds of different chemicals. Particles originate from sources and are also formed through atmospheric chemical reactions; the former are often referred to as “primary” particles, and the latter as “secondary” particles. In addition, there are also physical, non-chemical reaction mechanisms that contribute to secondary particles. Particle pollution also varies by time of year and location and is affected by several weather-related factors, such as temperature, clouds, humidity, and wind. A further layer of complexity comes from a particle’s ability to shift between solid/liquid and gaseous phases, which is influenced by concentration, meteorology, and temperature.

Fine particles are produced primarily by combustion processes and by transformations of gaseous emissions (e.g., NO_X, SO_X and VOCs) in the atmosphere. The chemical and physical properties of $PM_{2.5}$ may vary greatly with time, region, meteorology, and source

category. Thus, $PM_{2.5}$ may include a complex mixture of different pollutants including sulfates, nitrates, organic compounds, elemental carbon and metal compounds. These particles can remain in the atmosphere for days to weeks and travel through the atmosphere hundreds to thousands of kilometers.[1]

3.1.1.2 Ozone

Ground-level ozone pollution is formed by the reaction of VOCs and NO_X in the atmosphere in the presence of heat and sunlight. These pollutants, often referred to as ozone precursors, are emitted by many types of pollution sources such as highway vehicles and nonroad engines (including ships), power plants, chemical plants, refineries, makers of consumer and commercial products, industrial facilities, and smaller area sources.

The science of ozone formation, transport, and accumulation is complex.[2] Ground-level ozone is produced and destroyed in a cyclical set of chemical reactions, many of which are sensitive to temperature and sunlight. When ambient temperatures and sunlight levels remain high for several days and the air is relatively stagnant, ozone and its precursors can build up and result in more ozone than typically would occur on a single high-temperature day. Ozone can be transported hundreds of miles downwind of precursor emissions, resulting in elevated ozone levels even in areas with low VOC or NO_X emissions.

The highest levels of ozone are produced when both VOC and NO_X emissions are present in significant quantities on clear summer days. Relatively small amounts of NO_X enable ozone to form rapidly when VOC levels are relatively high, but ozone production is quickly limited by removal of the NO_X. Under these conditions NO_X reductions are highly effective in reducing ozone while VOC reductions have little effect. Such conditions are called "NO_X-limited." Because the contribution of VOC emissions from biogenic (natural) sources to local ambient ozone concentrations can be significant, even some areas where man-made VOC emissions are relatively low can be NO_X-limited.

Ozone concentrations in an area also can be lowered by the reaction of nitric oxide (NO) with ozone, forming nitrogen dioxide (NO_2); as the air moves downwind and the cycle continues, the NO_2 forms additional ozone. The importance of this reaction depends, in part, on the relative concentrations of NO_X, VOC, and ozone, all of which change with time and location. When NO_X levels are relatively high and VOC levels relatively low, NO_X forms inorganic nitrates (i.e., particles) but relatively little ozone. Such conditions are called "VOC-limited". Under these conditions, VOC reductions are effective in reducing ozone, but NO_X reductions can actually increase local ozone under certain circumstances. Even in VOC-limited urban areas, NO_X reductions are not expected to increase ozone levels if the NO_X reductions are sufficiently large.

Rural areas are usually NO_X-limited, due to the relatively large amounts of biogenic VOC emissions in such areas. Urban areas can be either VOC- or NO_X-limited, or a mixture of both, in which ozone levels exhibit moderate sensitivity to changes in either pollutant.

3.1.1.3 NO_X and SO_X

Sulfur dioxide (SO_2), a member of the sulfur oxide (SO_X) family of gases, is formed from burning fuels containing sulfur (e.g., coal or oil), extracting gasoline from oil, or extracting metals from ore. Nitrogen dioxide (NO_2) is a member of the nitrogen oxide (NO_X) family of gases. Most NO_2 is formed in the air through the oxidation of nitric oxide (NO) emitted when fuel is burned at a high temperature.

SO_2 and NO_2 can dissolve in water vapor and further oxidize to form sulfuric and nitric acid which reacts with ammonia to form sulfates and nitrates, both of which are important components of ambient PM. The health effects of ambient PM are discussed in Section 3.1.2.1. NO_X along with non-methane hydrocarbons (NMHC) are the two major precursors of ozone. The health effects of ozone are covered in Section 3.1.2.2.

3.1.1.4 Diesel Exhaust PM

Ship emissions contribute to ambient levels of air toxics known or suspected as human or animal carcinogens, or that have noncancer health effects. The population experiences an elevated risk of cancer and other noncancer health effects from exposure to air toxics.[3] These compounds include diesel PM.

Marine diesel engines emit diesel exhaust (DE), a complex mixture comprised of carbon dioxide, oxygen, nitrogen, water vapor, carbon monoxide, nitrogen compounds, sulfur compounds and numerous low molecular-weight hydrocarbons. A number of these gaseous hydrocarbon components are individually known to be toxic including aldehydes, benzene and 1,3-butadiene. The diesel particulate matter (DPM) present in diesel exhaust consists of fine particles (< 2.5μm), including a subgroup with a large number of ultrafine particles (< 0.1 μm). These particles have a large surface area which makes them an excellent medium for adsorbing organics, and their small size makes them highly respirable. Many of the organic compounds present in the gases and on the particles, such as polycyclic organic matter (POM), are individually known to have mutagenic and carcinogenic properties. Marine diesel engine emissions consist of a higher fraction of hydrated sulfate (approximately 60-90%) due to the higher sulfur levels of the fuel, organic carbon (approximately 15-30%), and metallic ash (approximately 7-11%) than are typically found in land-based engines.[4] In addition, while toxic trace metals emitted by marine diesel engines represent a very small portion of the national emissions of metals (less than one percent) and are a small portion of DPM (generally much less than one percent of DPM), we note that several trace metals of potential toxicological significance and persistence in the environment are emitted by diesel engines.[5] These trace metals include chromium, manganese, mercury, and nickel. In addition, small amounts of dioxins have been measured in highway engine diesel exhaust, some of which may partition into the particulate phase. Dioxins are a major health concern but diesel engines are a minor contributor to overall dioxin emissions.

Diesel exhaust varies significantly in chemical composition and particle sizes between different engine types (heavy-duty, light-duty), engine operating conditions (idle, accelerate, decelerate), and fuel formulations (high/low sulfur fuel). Also, there are emissions differences between on-road and nonroad engines because the nonroad engines are generally

of older technology. This is especially true for marine diesel engines.[6] After being emitted in the engine exhaust, diesel exhaust undergoes dilution as well as chemical and physical changes in the atmosphere. The lifetime for some of the compounds present in diesel exhaust ranges from hours to days.

3.1.2 Health Effects Associated with Exposure to Pollutants

3.1.2.1 PM Health Effects

This section provides a summary of the health effects associated with exposure to ambient concentrations of PM.[A] The information in this section is based on the data and conclusions in the PM Air Quality Criteria Document (PM AQCD) and PM Staff Paper prepared by the U.S. Environmental Protection Agency (EPA).[B,7,8] We also present additional recent studies published after the cut-off date for the PM AQCD.[9,C] Taken together this information supports the conclusion that exposure to ambient concentrations of PM are associated with adverse health effects. Information specifically related to health effects associated with exposure to diesel exhaust PM is included in Section 3.1.2.5 of this document.

3.1.2.1.1 Short-term Exposure Mortality and Morbidity Studies

As discussed in the PM AQCD, short-term exposure to $PM_{2.5}$ is associated with premature mortality from cardiopulmonary diseases,[10] hospitalization and emergency department visits for cardiopulmonary diseases,[11] increased respiratory symptoms,[12] decreased lung function[13] and physiological changes or biomarkers for cardiac changes.[14] In addition, the PM AQCD described a limited body of new evidence from epidemiologic

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[A] Personal exposure includes contributions from many different types of particles, from many sources, and in many different environments. Total personal exposure to PM includes both ambient and nonambient components; and both components may contribute to adverse health effects.

[B] The PM NAAQS is currently under review and the EPA is considering all available science on PM health effects, including information which has been published since 2004, in the development of the upcoming PM Integrated Science Assessment Document (ISA). A first draft of the PM ISA was completed in December 2008 and was submitted for review by the Clean Air Scientific Advisory Committee (CASAC) of EPA's Science Advisory Board. Comments from the general public have also been requested. For more information, see http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=201805.

[C] These additional studies are included in the 2006 Provisional Assessment of Recent Studies on Health Effects of Particulate Matter Exposure. The provisional assessment did not and could not (given a very short timeframe) undergo the extensive critical review by CASAC and the public, as did the PM AQCD. The provisional assessment found that the "new" studies expand the scientific information and provide important insights on the relationship between PM exposure and health effects of PM. The provisional assessment also found that "new" studies generally strengthen the evidence that acute and chronic exposure to fine particles and acute exposure to thoracic coarse particles are associated with health effects. Further, the provisional science assessment found that the results reported in the studies did not dramatically diverge from previous findings, and taken in context with the findings of the CD, the new information and findings did not materially change any of the broad scientific conclusions regarding the health effects of PM exposure made in the CD. However, it is important to note that this assessment was limited to screening, surveying, and preparing a provisional assessment of these studies. For reasons outlined in Section I.C of the preamble for the final PM NAAQS rulemaking in 2006 (see 71 FR 61148-49, October 17, 2006), EPA based its decision on the science presented in the 2004 CD.

studies for potential relationships between short term exposure to PM and health endpoints such as low birth weight, preterm birth, and neonatal and infant mortality.[15]

Among the studies of effects associated with short-term exposure to $PM_{2.5}$, several specifically address the contribution of mobile sources to short-term $PM_{2.5}$-related effects on premature mortality. The results from these studies generally indicated that several combustion-related fine particle source-types are likely associated with mortality, including motor vehicle emissions as well as other sources.[16] The analyses incorporate source apportionment tools into short-term exposure studies and are briefly mentioned here. Analyses incorporating source apportionment by factor analysis with daily time-series studies of daily death rates indicated a relationship between mobile source $PM_{2.5}$ and mortality.[17,18,19,20] Another recent study in 14 U.S. cities examined the effect of PM_{10} exposures on daily hospital admissions for cardiovascular disease. This study found that the effect of PM_{10} was significantly greater in areas with a larger proportion of PM_{10} coming from motor vehicles, indicating that PM_{10} from these sources may have a greater effect on the toxicity of ambient PM_{10} when compared with other sources.[21] These studies provide evidence that PM-related emissions, specifically from mobile sources, are associated with adverse health effects.

3.1.2.1.2 Long-term Exposure Mortality and Morbidity Studies

Long-term exposure to ambient $PM_{2.5}$ is associated with premature mortality from cardiopulmonary diseases and lung cancer,[22] and effects on the respiratory system such as decreased lung function or the development of chronic respiratory disease.[23] Of specific importance, the PM AQCD also noted that the PM components of gasoline and diesel engine exhaust represent one class of hypothesized likely important contributors to the observed ambient PM-related increases in lung cancer incidence and mortality.[24]

The PM AQCD and PM Staff Paper emphasized the results of two long-term epidemiologic studies, the Six Cities and American Cancer Society (ACS) prospective cohort studies, based on several factors – the large air quality data set for PM in the Six Cities Study, the fact that the study populations were similar to the general population, and the fact that these studies have undergone extensive reanalysis.[25,26,27,28 29,30] These studies indicate that there are positive associations for all-cause, cardiopulmonary, and lung cancer mortality with long-term exposure to $PM_{2.5}$. One analysis of a subset of the ACS cohort data, which was published after the PM AQCD was finalized but in time for the 2006 Provisional Assessment, found a larger association than had previously been reported between long-term $PM_{2.5}$ exposure and mortality in the Los Angeles area using a new exposure estimation method that accounted for variations in concentration within the city.[31]

As discussed in the PM AQCD, the morbidity studies that combine the features of cross-sectional and cohort studies provide the best evidence for chronic exposure effects. Long-term studies evaluating the effect of ambient PM on children's development have shown some evidence indicating effects of $PM_{2.5}$ and/or PM_{10} on reduced lung function growth.[32] In another recent publication included in the 2006 Provisional Assessment, investigators in southern California reported the results of a cross-sectional study of outdoor $PM_{2.5}$ and a measure of atherosclerosis development in the Los Angeles basin.[33] The study

found significant associations between ambient residential $PM_{2.5}$ and carotid intima-media thickness (CIMT), an indicator of subclinical atherosclerosis, an underlying factor in cardiovascular disease.

3.1.2.2 Ozone Health Effects

This section provides a summary of the health effects associated with ambient ozone.[D] The information in this section is based on the data and conclusions in the ozone air quality criteria document (ozone AQCD) and ozone staff paper prepared by the U.S. EPA.[34,35] Taken together this information supports the conclusion that ozone-related emissions are associated with adverse health effects.

Ozone-related health effects include lung function decrements, respiratory symptoms, aggravation of asthma, increased hospital and emergency room visits, increased asthma medication usage, and a variety of other respiratory effects. Cellular-level effects, such as inflammation of lungs, have been documented as well. In addition, there is suggestive evidence of a contribution of ozone to cardiovascular-related morbidity and highly suggestive evidence that short-term ozone exposure directly or indirectly contributes to non-accidental and cardiopulmonary-related mortality, but additional research is needed to clarify the underlying mechanisms causing these effects. In a recent report on the estimation of ozone-related premature mortality published by the National Research Council (NRC), a panel of experts and reviewers concluded that short-term exposure to ambient ozone is likely to contribute to premature deaths and that ozone-related mortality should be included in estimates of the health benefits of reducing ozone exposure.[36] People who appear to be more susceptible to effects associated with exposure to ozone include children, asthmatics and the elderly. Those with greater exposures to ozone, for instance due to time spent outdoors (e.g., children and outdoor workers), are also of concern.

A large number of scientific studies have identified several key health effects associated with exposure to levels of ozone found today in many areas of the United States. Short-term (1 to 3 hours) and prolonged exposures (6 to 8 hours) to ambient ozone concentrations have been linked to lung function decrements, respiratory symptoms, increased hospital admissions and emergency room visits for respiratory problems.[37, 38, 39, 40, 41, 42] Repeated exposure to ozone can increase susceptibility to respiratory infection and lung inflammation and can aggravate preexisting respiratory diseases, such as asthma.[43, 44, 45, 46, 47] Repeated exposure to sufficient concentrations of ozone can also cause inflammation of the lung, impairment of lung defense mechanisms, and possibly irreversible changes in lung structure, which over time could affect premature aging of the lungs and/or the development of chronic respiratory illnesses, such as emphysema and chronic bronchitis.[48, 49, 50, 51]

Children and adults who are outdoors and active during the summer months, such as

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[D] Human exposure to ozone varies over time due to changes in ambient ozone concentration and because people move between locations which have notable different ozone concentrations. Also, the amount of ozone delivered to the lung is not only influenced by the ambient concentrations but also by the individuals breathing route and rate.

construction workers, are among those most at risk of elevated ozone exposures. [52] Children and outdoor workers tend to have higher ozone exposure because they typically are active outside, working, playing and exercising, during times of day and seasons (e.g., the summer) when ozone levels are highest. [53] For example, summer camp studies in the Eastern United States and Southeastern Canada have reported statistically significant reductions in lung function in children who are active outdoors.[54, 55, 56, 57, 58, 59, 60, 61] Further, children are more at risk of experiencing health effects from ozone exposure than adults because their respiratory systems are still developing. These individuals (as well as people with respiratory illnesses, such as asthma, especially asthmatic children) can experience reduced lung function and increased respiratory symptoms, such as chest pain and cough, when exposed to relatively low ozone levels during prolonged periods of moderate exertion.[62, 63, 64, 65]

3.1.2.3 SO_X Health Effects

This section provides an overview of the health effects associated with SO_2. Additional information on the health effects of SO_2 can be found in the U.S. Environmental Protection Agency Integrated Science Assessment for Sulfur Oxides.[66] Following an extensive evaluation of health evidence from epidemiologic and laboratory studies, the U.S. EPA has concluded that there is a causal relationship between respiratory health effects and short-term exposure to SO_2. The immediate effect of SO_2 on the respiratory system in humans is bronchoconstriction. This response is mediated by chemosensitive receptors in the tracheobronchial tree. These receptors trigger reflexes at the central nervous system level resulting in bronchoconstriction, mucus secretion, mucosal vasodilation, cough, and apnea followed by rapid shallow breathing. In some cases, local nervous system reflexes also may be involved. Asthmatics are more sensitive to the effects of SO_2 likely resulting from preexisting inflammation associated with this disease. This inflammation may lead to enhanced release of mediators, alterations in the autonomic nervous system and/or sensitization of the chemosensitive receptors. These biological processes are likely to underlie the bronchoconstriction and decreased lung function observed in response to SO_2 exposure. In laboratory studies involving controlled human exposures to SO_2, respiratory effects have consistently been observed following 5-10 min exposures at SO_2 concentrations $\geq$ 0.2 ppm in asthmatics engaged in moderate to heavy levels of exercise. In these studies, 5-30% of relatively healthy exercising asthmatics are shown to experience moderate or greater decrements in lung function ($\geq$ 100% increase in sRaw (specific airway resistance) or $\geq$ 15% decrease in FEV_1 (forced expiratory volume in 1 second)) with peak exposures to SO_2 concentrations of 0.2-0.3 ppm. At concentrations $\geq$ 0.4 ppm, a greater percentage of asthmatics (20-60%) experience SO_2-induced decrements in lung function, which are frequently accompanied by respiratory symptoms. A clear concentration-response relationship has been demonstrated in laboratory studies following exposures to SO_2 at concentrations between 0.2 and 1.0 ppm, both in terms of increasing severity of effect and percentage of asthmatics adversely affected.

In epidemiologic studies, respiratory effects have been observed in areas where the mean 24-hour SO_2 levels range from 1 to 30 ppb, with maximum 1 to 24-hour average SO_2 values ranging from 12 to 75 ppb. Important new multicity studies and several other studies have found an association between 24-hour average ambient SO_2 concentrations and respiratory symptoms in children, particularly those with asthma. Furthermore, limited

epidemiologic evidence indicates that atopic children and adults may be at increased risk for SO_2-induced respiratory symptoms. Generally consistent associations also have been observed between ambient SO_2 concentrations and emergency department visits and hospitalizations for all respiratory causes, particularly among children and older adults ($\geq$ 65 years), and for asthma. Intervention studies provide additional evidence that supports a causal relationship between SO_2 exposure and respiratory health effects. Two notable studies conducted in several cities in Germany and in Hong Kong reported that decreases in SO_2 concentrations were associated with improvements in respiratory symptoms, though the possibility remained that these health improvements may be partially attributable to declining concentrations of air pollutants other than SO_2, most notably PM or constituents of PM. A limited subset of epidemiologic studies has examined potential confounding by copollutants using multipollutant regression models. These analyses indicate that although copollutant adjustment has varying degrees of influence on the SO_2 effect estimates, the effect of SO_2 on respiratory health outcomes appears to be generally robust and independent of the effects of gaseous and particulate copollutants, suggesting that the observed effects of SO_2 on respiratory endpoints occur independent of the effects of other ambient air pollutants.

Consistent associations between short-term exposure to SO_2 and mortality have been observed in epidemiologic studies, with larger effect estimates reported for respiratory mortality than cardiovascular mortality. While this finding is consistent with the demonstrated effects of SO_2 on respiratory morbidity, uncertainty remains with respect to the interpretation of these associations due to potential confounding by various copollutants. The U.S. EPA has therefore concluded that the overall evidence is suggestive of a causal relationship between short-term exposure to SO_2 and mortality. Significant associations between short-term exposure to SO_2 and emergency department visits and hospital admissions for cardiovascular diseases have also been reported. However, these findings have been inconsistent across studies and do not provide adequate evidence to infer a causal relationship between SO_2 exposure and cardiovascular morbidity.

3.1.2.4 NO_X Health Effects

This section provides an overview of the health effects associated with NO_2. Additional information on the health effects of NO_2 can be found in the U.S. Environmental Protection Agency Integrated Science Assessment (ISA) for Nitrogen Oxides.[67] The U.S. EPA has concluded that the findings of epidemiologic, controlled human exposure, and animal toxicological studies provide evidence that is sufficient to infer a likely causal relationship between respiratory effects and short-term NO_2 exposure. [68] The ISA concludes that the strongest evidence for such a relationship comes from epidemiologic studies of respiratory effects including symptoms, emergency department visits, and hospital admissions. [69] The effect estimates from U.S. and Canadian studies generally indicate that ambient NO_2 is associated with a 2-20% increase in risks for emergency department visits and hospital admissions. Risks associated with respiratory symptoms are generally higher.[70] These epidemiologic studies are supported by evidence from experimental studies, in particular by controlled human exposure studies that evaluate airway hyperresponsiveness in asthmatic individuals.[71] The ISA draws two broad conclusions regarding airway responsiveness following NO_2 exposure.[72] First, the ISA concludes that NO2 exposure may enhance the sensitivity to allergen-induced decrements in lung function and increase the

allergen-induced airway inflammatory response at exposures as low as 0.26 ppm NO_2 for 30 minutes.[73] Second, exposure to NO_2 has been found to enhance the inherent responsiveness of the airway to subsequent nonspecific challenges in controlled human exposure studies.[74] In general, small but significant increases in nonspecific airway responsiveness were observed in the range of 0.2 to 0.3 ppm NO_2 for 30-minute exposures and at 0.1 ppm NO_2 for 60-minute exposures in asthmatics. These conclusions are consistent with results from animal toxicological studies which have detected 1) increased immume-mediated pulmonary inflammation in rats exposed to house dust mite allergen following exposure to 5 ppm NO_2 for 3-hour and 2) increased responsiveness to non-specific challenges following sub-chronic (6-12 weeks) exposure to 1 to 4 ppm NO_2.[75] Enhanced airway responsiveness could have important clinical implications for asthmatics since transient increases in airway responsiveness following NO_2 exposure have the potential to increase symptoms and worsen asthma control.[76] Together, the epidemiologic and experimental data sets form a plausible, consistent, and coherent description of a relationship between NO_2 exposures and an array of adverse health effects that range from the onset of respiratory symptoms to hospital admission.

Although the weight of evidence supporting a causal relationship is somewhat less certain than that associated with respiratory morbidity, NO_2 has also been linked to other health endpoints. For example, results from several large U.S. and European multi-city studies and a meta-analysis study indicate positive associations between ambient NO_2 concentrations and the risk of all-cause (nonaccidental) mortality, with effect estimates ranging from 0.5 to 3.6% excess risk in mortality per standardized increment (20 ppb for 24-hour averaging time, 30 ppb for 1-hour averaging time).[77] In general, the NO_2 effect estimates were robust to adjustment for co-pollutants. In addition, generally positive associations between short-term ambient NO_2 concentrations and hospital admissions or emergency department visits for cardiovascular disease have been reported.[78] A number of epidemiologic studies have also examined the effects of long-term exposure to NO_2 and reported positive associations with decrements in lung function and partially irreversible decrements in lung function growth.[79] Specifically, results from the California-based Children's Health Study, which evaluated NO_2 exposures in children over an 8-year period, demonstrated deficits in lung function growth. [80] This effect has also been observed in Mexico City, Mexico[81] and in Oslo, Norway, [82] with decrements ranging from 1 to 17.5 ml per 20- ppb increase in annual NO_2 concentration. Animal toxicological studies may provide biological plausibility for the chonic effects of NO_2 that have been observed in these epidemiologic studies.[83] The main biochemical targets of NO_2 exposure appear to be antioxidants, membrane polyunsaturated fatty acids, and thiol groups. NO_2 effects include changes in oxidant/antioxidant homeostasis and chemical alterations of lipids and proteins. Lipid peroxidation has been observed at NO_2 exposures as low as 0.04 ppm for 9 months and at exposures of 1.2 ppm for 1 week, suggesting lower effect thresholds with longer durations of exposure. Other studies showed decreases in formation of key arachidonic acid metabolites in mornings following NO_2 exposures of 0.5 ppm. NO_2 has been shown to increase collagen synthesis rates at concentrations as low as 0.5 ppm. This could indicate increased total lung collagen, which is associated with pulmonary fibrosis, or increased collagen turnover, which is associated with remodeling of lung connective tissue. Morphological effects following chonic NO_2 exposures have been identified in animal studies

that link to these increases in collagen synthesis and may provide plausibility for the deficits in lung function growth described in epidemiologic studies.[84]

3.1.2.5 Diesel Exhaust PM Health Effects

A large number of health studies have been conducted regarding diesel exhaust. These include epidemiologic studies of lung cancer in groups of workers and animal studies focusing on non-cancer effects. Diesel exhaust PM (including the associated organic compounds which are generally high molecular weight hydrocarbons but not the more volatile gaseous hydrocarbon compounds) is generally used as a surrogate exposure measure for whole diesel exhaust.

Diesel exhaust has been found to be of concern by several groups worldwide including the U.S. government. The IPCS (International Programme on Chemical Safety) has established an environmental health criteria for diesel fuel and exhaust emissions. In this criteria the IPCS recommends that for the protection of human health diesel exhaust emissions should be controlled. The IPCS explicitly states that urgent efforts should be made to reduce emissions, specifically of particulates, by changing exhaust train techniques, engine design and fuel composition.[85]

3.1.2.5.1 Potential Cancer Effects of Exposure to Diesel Exhaust

The U.S. EPA's 2002 final "Health Assessment Document for Diesel Engine Exhaust" (the EPA Diesel HAD) classified exposure to diesel exhaust as likely to be carcinogenic to humans by inhalation at environmental exposures, in accordance with the revised draft 1996/1999 U.S. EPA cancer guidelines.[86, 87] In accordance with earlier U.S. EPA guidelines, exposure to diesel exhaust would similarly be classified as probably carcinogenic to humans (Group B1).[88,89] A number of other agencies (National Institute for Occupational Safety and Health, the International Agency for Research on Cancer, the World Health Organization, California EPA, and the U.S. Department of Health and Human Services) have made similar classifications.[90, 91,92,93,94] The Health Effects Institute has prepared numerous studies and reports on the potential carcinogenicity of exposure to diesel exhaust.[95,96,97]

More specifically, the U.S. EPA Diesel HAD states that the conclusions of the document apply to diesel exhaust in use today including both onroad and nonroad engines including marine diesel engines present on ships. The U.S. EPA Diesel HAD acknowledges that the studies were done on engines with generally older technologies and that "there have been changes in the physical and chemical composition of some DE [diesel exhaust] emissions (onroad vehicle emissions) over time, though there is no definitive information to show that the emission changes portend significant toxicological changes." In any case, the diesel technology used for marine diesel engines typically lags that used for onroad engines which have been subject to PM standards since 1998. Thus it is reasonable to assume that the hazards identified from older technologies may be largely applicable to marine engines.

For the Diesel HAD, the U.S. EPA reviewed 22 epidemiologic studies on the subject of the carcinogenicity of exposure to diesel exhaust in various occupations, finding increased lung cancer risk, although not always statistically significant, in 8 out of 10 cohort studies and

10 out of 12 case-control studies which covered several industries. Relative risk for lung cancer, associated with exposure, ranged from 1.2 to 1.5, although a few studies show relative risks as high as 2.6. Additionally, the Diesel HAD also relied on two independent meta-analyses, which examined 23 and 30 occupational studies respectively, and found statistically significant increases of 1.33 to 1.47 in smoking-adjusted relative lung cancer risk associated with diesel exhaust. These meta-analyses demonstrate the effect of pooling many studies and in this case show the positive relationship between diesel exhaust exposure and lung cancer across a variety of diesel exhaust-exposed occupations.[98,99,100]

The U.S. EPA recently assessed air toxic emissions and their associated risk (the National-Scale Air Toxics Assessment or NATA for 1996 and 1999), and concluded that diesel exhaust ranks with other emissions that the national-scale assessment suggests pose the greatest relative risk.[101,102] This national assessment estimates average population inhalation exposures to DPM for nonroad and on-highway sources. These are the sum of ambient levels weighted by the amount of time people spend in each of the locations.

In summary, the likely hazard to humans together with the potential for significant environmental risks leads us to conclude that diesel exhaust emissions from marine engines present public health issues of concern.

3.1.2.5.2 Other Health Effects of Exposure to Diesel Exhaust

Noncancer health effects of acute and chronic exposure to diesel exhaust emissions are also of concern. The Diesel HAD established an inhalation Reference Concentration (RfC) specifically based on animal studies of diesel exhaust exposure. An RfC is defined by the U.S. EPA as "an estimate of a continuous inhalation exposure to the human population, including sensitive subgroups, with uncertainty spanning perhaps an order of magnitude, which is likely to be without appreciable risks of deleterious noncancer effects during a lifetime." The U.S. EPA derived the RfC from consideration of four well-conducted chronic rat inhalation studies showing adverse pulmonary effects.[103,104,105,106] The diesel RfC is based on a "no observable adverse effect" level of 144 $\mu g/m^3$ that is further reduced by applying uncertainty factors of 3 for interspecies extrapolation and 10 for human variations in sensitivity. The resulting RfC derived in the Diesel HAD is 5 $\mu g/m^3$ for diesel exhaust, as measured by DPM. This RfC does not consider allergenic effects such as those associated with asthma or immunologic effects. There is growing evidence that exposure to diesel exhaust can exacerbate these effects, but the exposure-response data is presently lacking to derive an RfC. The Diesel HAD states, "With DPM [diesel particulate matter] being a ubiquitous component of ambient PM, there is an uncertainty about the adequacy of the existing DE [diesel exhaust] noncancer database to identify all of the pertinent DE-caused noncancer health hazards" (p. 9-19).

While there have been relatively few human studies associated specifically with the noncancer impact of exposure to DPM alone, DPM is a component of the ambient particles studied in numerous epidemiologic studies. The conclusion that health effects associated with ambient PM in general are relevant to DPM is supported by studies that specifically associate observable human noncancer health effects with exposure to DPM. As described in the Diesel HAD, these studies identified some of the same health effects reported for ambient

PM, such as respiratory symptoms (cough, labored breathing, chest tightness, wheezing), and chronic respiratory disease (cough, phlegm, chronic bronchitis and suggestive evidence for decreases in pulmonary function). Symptoms of immunological effects such as wheezing and increased allergenicity are also seen. Studies in rodents, especially rats, show the potential for human inflammatory effects in the lung and consequential lung tissue damage from chronic diesel exhaust inhalation exposure. The Diesel HAD concludes "that acute exposure to DE [diesel exhaust] has been associated with irritation of the eye, nose, and throat, respiratory symptoms (cough and phlegm), and neurophysiological symptoms such as headache, lightheadedness, nausea, vomiting, and numbness or tingling of the extremities."[107] There is also evidence for an immunologic effect such as the exacerbation of allergenic responses to known allergens and asthma-like symptoms.[108,109,110]

The Diesel HAD briefly summarizes health effects associated with ambient PM and discusses the $PM_{2.5}$ NAAQS. There is a much more extensive body of human data, which is also mentioned earlier in the health effects discussion for $PM_{2.5}$ (Section 3.2.1.1 of this document), showing a wide spectrum of adverse health effects associated with exposure to ambient PM, of which diesel exhaust is an important component. The $PM_{2.5}$ NAAQS is designed to provide protection from the non-cancer and premature mortality effects of $PM_{2.5}$ as a whole.

3.1.2.5.3 Exposure to Diesel Exhaust PM

Exposure of people to diesel exhaust depends on their various activities, the time spent in those activities, the locations where these activities occur, and the levels of diesel exhaust pollutants in those locations. The major difference between ambient levels of diesel particulate and exposure levels for diesel particulate is that exposure levels account for a person moving from location to location, the proximity to the emission source, and whether the exposure occurs in an enclosed environment.

Occupational exposures to diesel exhaust from mobile sources, including marine diesel engines, can be several orders of magnitude greater than typical exposures in the non-occupationally exposed population. Over the years, diesel particulate exposures have been measured for a number of occupational groups resulting in a wide range of exposures from 2 to 1280 $\mu g/m^3$ for a variety of occupations. As discussed in the Diesel HAD, the National Institute of Occupational Safety and Health (NIOSH) has estimated a total of 1,400,000 workers are occupationally exposed to diesel exhaust from on-road and nonroad vehicles including marine diesel engines.

3.1.2.5.3.1 Elevated Concentrations and Ambient Exposures in Mobile Source-Impacted Areas

While occupational studies indicate that those working in closest proximity to diesel exhaust experience the greatest health effects, recent studies are showing that human populations living near large diesel emission sources such as major roadways, [111] rail yards, [112] and marine ports[113] are also likely to experience greater exposure to PM and other components of diesel exhaust than the overall population, putting them at a greater health risk.

The percentage of total port emissions that come from ships varies by port. However, ships contribute to the DPM concentrations at ports, and elsewhere, that influence exposures.

Regions immediately downwind of marine ports may experience elevated ambient concentrations of directly-emitted $PM_{2.5}$ from diesel engines. Due to the nature of marine ports, emissions from a large number of diesel engines are concentrated in a small area. A recent study from the California Air Resources Board (CARB) evaluated air quality impacts of diesel engine emissions within the Port of Long Beach and Los Angeles in California, one of the largest ports in the U.S.[114] The port study employed the ISCST3 dispersion model. With local meteorological data used in the modeling , annual average concentrations of DPM were substantially elevated over an area exceeding 200,000 acres. Because the Ports are located near heavily-populated areas, the modeling indicated that over 700,000 people lived in areas with at least 0.3 $\mu g/m^3$ of port-related DPM in ambient air, about 360,000 people lived in areas with at least 0.6 $\mu g/m^3$ of DPM, and about 50,000 people lived in areas with at least 1.5 $\mu g/m^3$ of ambient DPM emitted directly from the port. This port study highlights the substantial contribution these facilities make to ambient concentrations of DPM in large, densely populated areas.

Figure 3.1-1 provides an aerial shot of the Port of Long Beach and Los Angeles in California.

Figure 3.1-1 Aerial Shot – Port of LA and Long Beach, California

The U.S. EPA recently updated its initial screening-level analysis[115,116] of selected marine port areas to better understand the populations, including minority, low-income, and

children, that are exposed to diesel particulate matter (DPM) emissions from these facilities.[E] The results of this study are discussed here and are also available in the public docket.[117,118]

This screening-level analysis focused on a representative selection of national marine ports.[F] Of the 45 marine ports studied, the results indicate that at least 18 million people, including a disproportionate number of low-income households, African-Americans, and Hispanics, live in the vicinity of these facilities and are being exposed to ambient DPM levels that are 2.0 $\mu g/m^3$ and 0.2 $\mu g/m^3$ above levels found in areas further from these facilities. Considering only ocean-going marine engine DPM emissions, the results indicate that 6.5 million people are exposed to ambient DPM levels that are 2.0 $\mu g/m^3$ and 0.2 $\mu g/m^3$ above levels found in areas further from these facilities. Because those populations exposed to DPM emissions from marine ports are more likely to be low-income and minority residents, these populations would benefit from the standards being proposed by the coordinated strategy. The detailed findings of this study are available in the public docket.

With regard to children, this analysis shows that at least four million children live in the vicinity of the marine ports studied and are also exposed to ambient DPM levels that are 2.0 $\mu g/m^3$ and 0.2 $\mu g/m^3$ above levels found in areas further from these facilities. Of the 6.5 million people exposed to DPM emissions from ocean-going vessel emissions, 1.7 million are children. The age composition of the total affected population in the screening analysis matches closely with the age composition of the overall US population. However, for some individual facilities the young (0-4 years) appear to be over-represented in the affected population compared to the overall US population. Detailed results for individual harbors are presented in the Appendices of the memorandum in the docket.

As part of this study, a computer geographic information system was used to identify the locations and boundaries of a the harbor areas, and determine the size and demographic characteristics of the populations living near these facilities. These facilities are listed in Table 3.1-1. Figures 3.1-2 and 3.1-3 provide examples of digitized footprints of the marine harbor areas included in this study.

[E] This type of screening-level analysis is an inexact tool and not appropriate for regulatory decision-making; it is useful in beginning to understand potential impacts and for illustrative purposes.

[F] The Agency selected a representative sample from the top 150 U.S. ports including coastal, inland, and Great Lake ports.

Table 3.1-1 Marine Harbor Areas

Baltimore, MD	Los Angeles, CA	Port of Baton Rouge, LA
Boston, MA	Louisville, KY	Port of Plaquemines, LA
Charleston, SC	Miami, FL	Portland, ME
Chicago, IL	Mobile, AL	Portland, OR
Cincinnati, OH	Mount Vernon, IN	Richmond, CA
Cleveland, OH	Nashville, TN	Savannah, GA
Corpus Christi, TX	New Orleans, LA	Seattle, WA
Detroit, MI	New York, NY	South Louisiana, LA
Duluth-Superior, MN	Oakland, CA	St. Louis, MO
Freeport, TX	Panama City, FL	Tacoma, WA
Gary, IN	Paulsboro, NJ	Tampa, FL
Helena, AR	Philadelphia, PA	Texas City, TX
Houston, TX	Pittsburgh, PA	Tulsa - Port of Catoosa, OK
Lake Charles, LA	Port Arthur, TX	Two Harbors, MN
Long Beach, CA	Port Everglades, FL	Wilmington, NC

Figure 3.1-2 Digitized footprint of New York, NY harbor area.

Figure 3.1-3 Digitized footprint of Portland, OR harbor area.

In order to better understand the populations that live in the vicinity of marine harbor areas and their potential exposures to ambient DPM, concentration isopleths surrounding the 45 marine port areas were created and digitized for all emission sources at the marine port and for ocean-going vessel Category 3 engine emissions only. The concentration isopleths of interest were selected to correspond to two DPM concentrations above urban background, 2.0 $\mu g/m^3$ and 0.2 $\mu g/m^3$. The isopleths were estimated using the AERMOD air dispersion model. Figures 3.1-4 and 3.1-5 provide examples of concentration isopleths surrounding the New York, NY harbor area for all emission sources and for ocean-going vessel Category 3 only engine emissions, respectively.

Figure 3.1-4 Concentration isopleths of New York, NY harbor area resulting from all emission sources.

Figure 3.1-5 Concentration isopleths of New York, NY harbor area resulting from only Category 3 vessels.

The size and characteristics of populations and households that reside within the area encompassed by the two DPM concentration isopleths were determined for each isopleth and the demographic compositions were assessed, including age, income level, and race/ethnicity.

In summary, the screening-level analysis found that for the 45 U.S. marine ports studied, al least 18 million people live in the vicinity of these facilities and are exposed to ambient DPM levels from all port emission sources that are 2.0 $\mu g/m^3$ and 0.2 $\mu g/m^3$ above those found in areas further from these facilities. If only Category 3 engine DPM emissions are considered, then the number of people exposed is 6.5 million.

3.1.2.6 Alaska and Hawaii Health Effects

The U.S. air quality maps below do not show Alaska and Hawaii. This is because the domain of the CMAQ model does not include these states. However ship emission inventories for Alaska and Hawaii were developed and are included in the totals presented in Section 7. Based on the inventory there are substantial ship emissions in the proposed ECA areas around Alaska and Hawaii. These are also the areas where most of the states' populations reside. Two of Alaska's three biggest population centers (Anchorage: population

260,000 and Juneau: population 30,000) are on the southeastern coast and these 2 cities alone are home to just under half of the entire state's population. In Hawaii, more than 99% of the state's population lives in the proposed ECA area. Meteorological information in Section 6 suggests that these emissions affect air quality. Based on Canadian air quality modeling, there would be significant air quality improvements for Eastern Alaska along the Canadian border. Therefore, it is reasonable to expect ships are contributing to ambient air concentrations of ozone and $PM_{2.5}$ in Hawaii and Alaska, even though our modeling does not allow us to quantify these effects.

3.2 Current and Projected Air Quality

Ships are currently contributing to ambient $PM_{2.5}$ and ozone concentrations and their contribution will continue to grow into the future as more stringent controls for onshore emission sources take effect. In this section, we present information on $PM_{2.5}$ and ozone levels in the continental United States based on air quality modeling. We also discuss the air quality modeling methodology and impacts from ships' emissions on air quality in Alaska and Hawaii.

Due to the imprecise science of discerning human health effects that are due solely to SO_X versus its PM derivatives (i.e. sulphate particles) or to NO_X versus its derivatives, ozone and PM, the air quality and health impacts from exposure to direct SO_X and NO_X from ships are not separately quantified here.

3.2.1 Current $PM_{2.5}$ Levels

As described in Section 3.1.2, PM causes adverse health effects, and the U.S. government has set national standards to protect against those health effects. There are two U.S. national ambient air quality standards (NAAQS) for $PM_{2.5}$: an annual standard (15 $\mu g/m^3$) and a 24-hour standard (35 $\mu g/m^3$). The most recent revisions to these standards were in 1997 and 2006. In 2005 the U.S. EPA designated nonattainment areas for the 2006 $PM_{2.5}$ NAAQS (70 FR 19844, April 14, 2005).[G]

In addition to the U.S. government NAAQS for $PM_{2.5}$, the World Health Organization (WHO) has also set air quality guidelines for $PM_{2.5}$.[119] The 2005 WHO Air Quality Guidelines (AQG) set for the first time a guideline value for particulate matter (PM). Although the aim is to achieve the lowest concentrations possible, since no threshold for PM has been identified below which no damage to health is observed, the annual mean $PM_{2.5}$ AQG is 10 $\mu g/m^3$ and the 24-hour mean $PM_{2.5}$ AQG is 25 $\mu g/m^3$.

The IMO, the U.S. government and individual states and local areas have already put in place many $PM_{2.5}$ and $PM_{2.5}$ precursor emission reduction programs. However, ships are significant contributors to $PM_{2.5}$ in many areas and states will need additional reductions in a timely manner to help them meet their air quality goals.

[G] A nonattainment area is defined in the Clean Air Act (CAA) as an area that is violating an ambient standard or is contributing to a nearby area that is violating the standard.

3.2.2 Projected $PM_{2.5}$ Air Quality

Levels of $PM_{2.5}$ in the ambient air are expected to continue to be a problem into the future. Without further action, emissions from ships will contribute a larger share to the projected levels of $PM_{2.5}$ as emissions from other sources decrease. In this section we present information on projected levels of $PM_{2.5}$ in 2020, ships' contribution to these levels, and the improvements which would occur with the proposed ECA.

3.2.2.1 Projected $PM_{2.5}$ Levels without an ECA

Figure 3.2-1 presents the projected annual average $PM_{2.5}$ concentrations for the continental U.S.[H] based on the inventory projections described in Section 2.7.[I] Most of the U.S. is projected to have annual average $PM_{2.5}$ levels between 5 and 12 $\mu g/m^3$ with a few areas having higher levels and some areas in the west having lower levels.

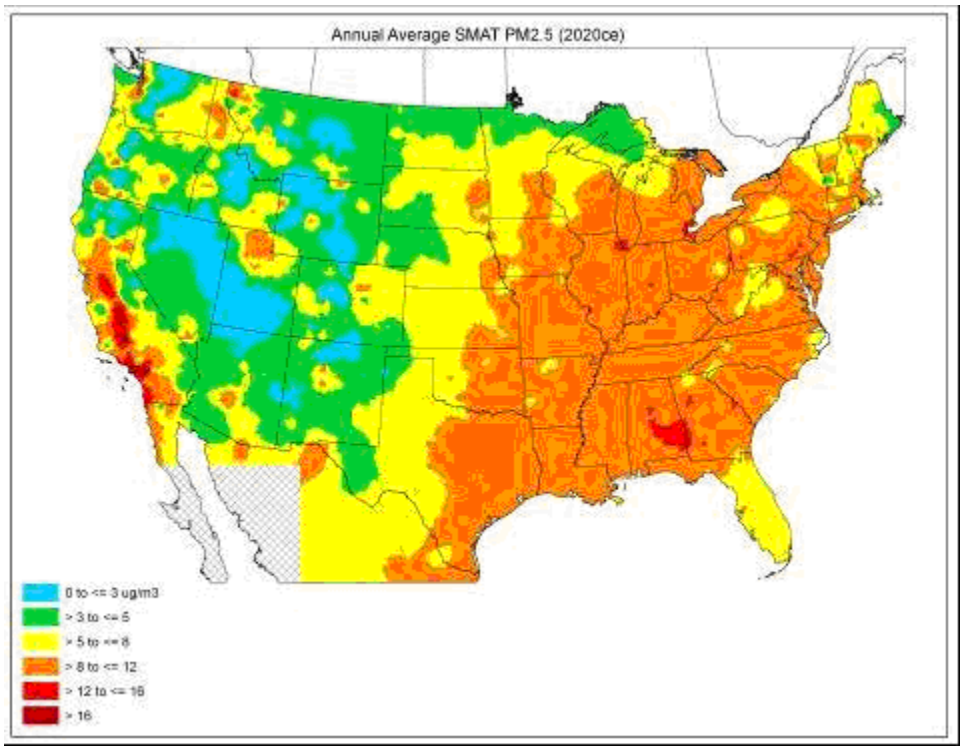

Figure 3.2-1 Annual Average $PM_{2.5}$ Concentrations in 2020 without an ECA

Even with the implementation of all current U.S. state and federal regulations, there are projected to be many areas in the U.S. with levels of $PM_{2.5}$ which are above health

[H] As discussed in Section 3.2.5.1.2 the air quality modeling domain only covers the continental United States.
[I] As discussed in Section 2.7 the inventories used for the air quality modeling differ slightly from those used in the final inventory calculations. The difference is small and was due to an error in calculating the distances and the fact that the air quality modeling only included Tier I NO_X controls in the baseline.

standards.[J] Emission reductions from the ECA designation will be helpful for states and counties in attaining and maintaining the $PM_{2.5}$ NAAQS and the WHO AQG.

3.2.2.2 Contribution of Ships to Projected $PM_{2.5}$ Levels

Emissions of NO_X, SO_X and direct $PM_{2.5}$ from ships have a significant impact on ambient $PM_{2.5}$ concentrations. The contribution from ships were determined by comparing model results in two future year control runs, one with all sources and one without ships. Figure 3.2-2 illustrates the projected percentage contribution of ships to annual average $PM_{2.5}$ concentrations in 2020. The percentage contribution of ships to annual average $PM_{2.5}$ concentrations is projected to be greater than 15% in parts of southern FL, southern LA, and the northern and southern Pacific coastline. The impact of ship emissions on $PM_{2.5}$ concentrations also extends well beyond the U.S. coastlines. As can be seen in Figure 3.2-2 the projected contribution of ships to annual average $PM_{2.5}$ concentrations in many inland areas, such as Tennessee, Nevada, New York and Pennsylvania, is up to 2%.

The absolute contribution of ships to ambient $PM_{2.5}$ levels is shown in Figure 3.2-3. This shows that the contribution from ships to annual average $PM_{2.5}$ concentrations is projected to be greater than 3 $\mu g/m^3$ for highly populated portions of southern California, while both southern Louisiana and Florida are projected to show impacts greater than 1.5 $\mu g/m^3$.

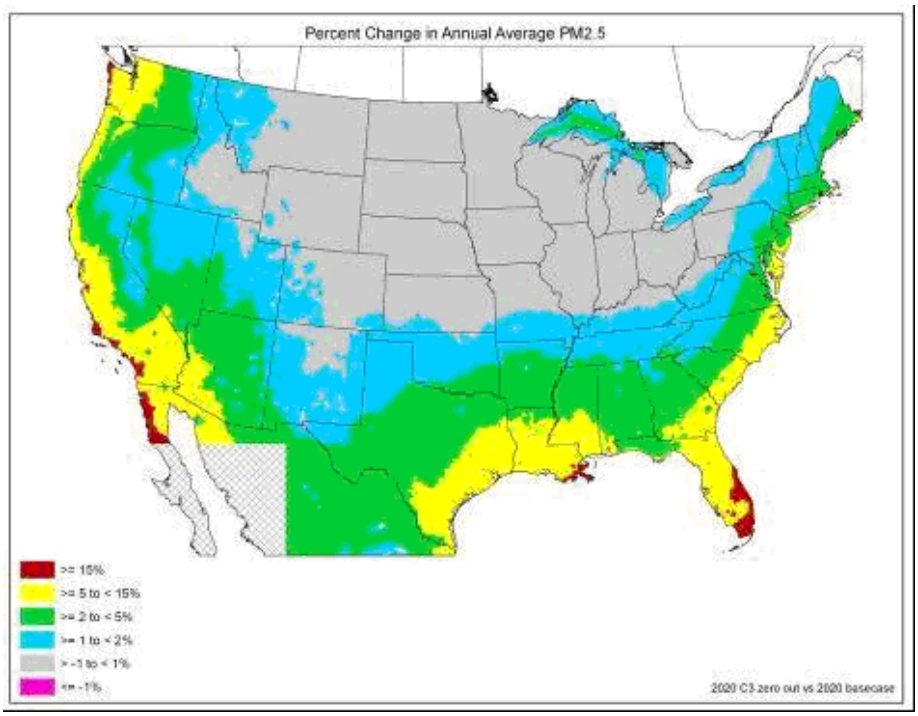

Figure 3.2-2 Percentage Contribution of Ships to Annual Average $PM_{2.5}$ Concentrations in 2020

[J] See Chapter 5, Section 5.4 for more information about existing emission reduction programs to control land-based and other marine sources.

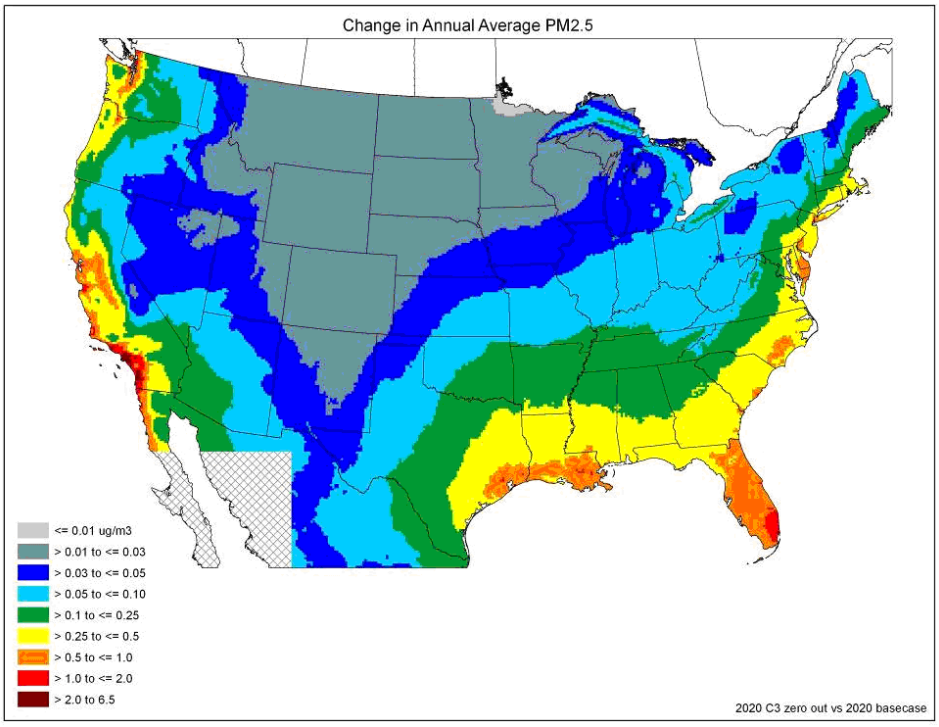

Figure 3.2-3 Absolute Contribution of Ships to Annual Average $PM_{2.5}$ Concentrations in 2020

The modeling projections clearly show that ships affect air quality far inland on all the U.S. coastlines. This is to be expected since ships operate along all the U.S. coastlines. It can be concluded from looking at these results that emissions from ships need to be controlled in order to achieve $PM_{2.5}$ reductions, even in inland areas and areas without ports.

3.2.2.3 Projected $PM_{2.5}$ Levels with an ECA

The impacts of the proposed ECA were determined by comparing the model results in the 2020 control run against the baseline simulation of the same year. According to air quality modeling performed for this analysis, the emission standards are expected to provide significant nationwide improvements in $PM_{2.5}$ levels.

Figures 3.2-4 and 3.2-5 present the projected percentage and absolute $PM_{2.5}$ improvements in 2020 if an ECA were enacted 200 nm from the U.S. shoreline. Similar to Figures 3.2-2 and 3.2-3, the $PM_{2.5}$ improvements extend well inland including southern California, the cities of Birmingham, AL and Atlanta, GA and the northeast corridor. The entire U.S. coastline will experience large improvements in their air quality from the proposed ECA.

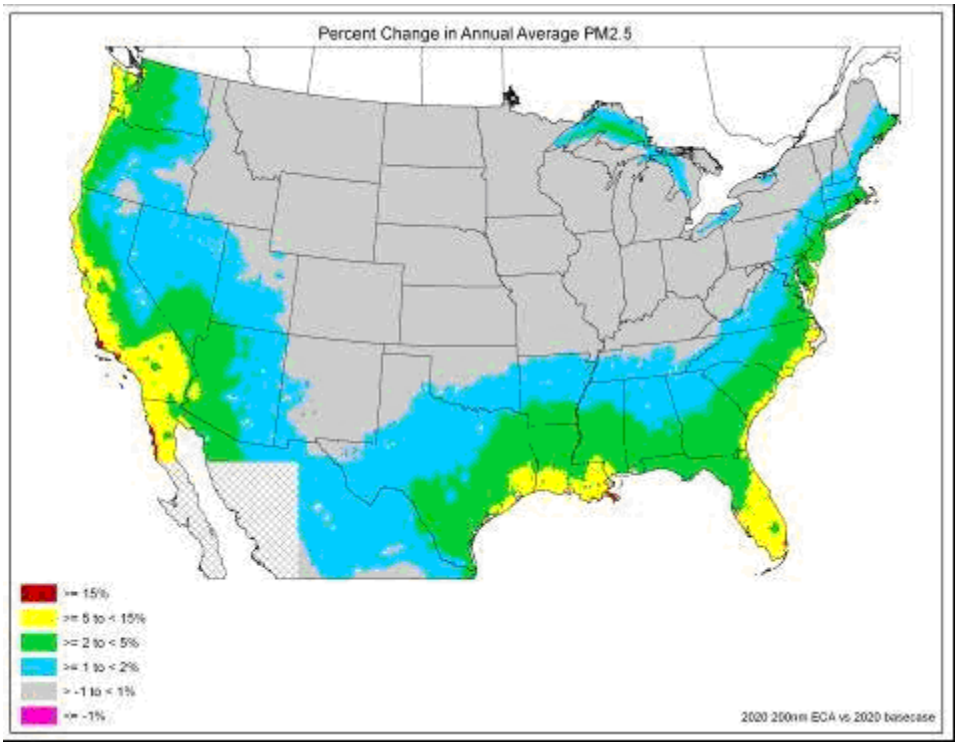

Figure 3.2-4 Percent Improvement in Annual Average $PM_{2.5}$ Concentrations in 2020 Resulting from the Application of the Proposed ECA

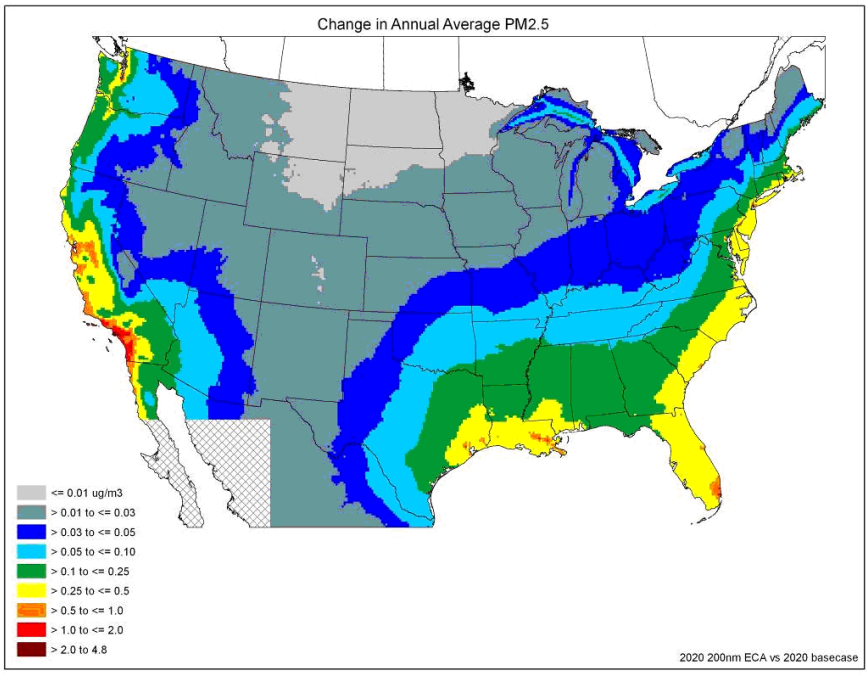

Figure 3.2-5 Absolute Improvement in Annual Average $PM_{2.5}$ Concentrations in 2020 Resulting from the Application of the Proposed ECA

3.2.3 Current Ozone Levels

As described in Section 3.1.2, ozone causes adverse health effects, and the U.S. government has set national standards to protect against those health effects. The U.S. EPA has recently amended the ozone NAAQS (73 FR 16436, March 27, 2008). The final 2008 ozone NAAQS rule addresses revisions to the previous 1997 NAAQS for ozone to provide increased protection of public health and welfare. The 1997 8-hour ozone NAAQS was set at 0.08 ppm (effectively 0.084 ppm). In 2008 the U.S. EPA revised the level of the 8-hour standard to 0.075 parts per million (ppm), expressed to three decimal places.

In addition to the U.S. government NAAQS for ozone, the WHO has also set an AQG for ozone of 100 $\mu g/m^3$ for an 8-hour mean.[120] Comparing the WHO AQG to the U.S. NAAQS requires converting $\mu g/m^3$ to ppb and assuming a temperature of 20º Celsius and an atmospheric pressure of 1013 mb. The conversion is approximately a factor of 2, meaning that the AQG for ozone is approximately 50 ppb.[K,121,122]

The IMO, the U.S. government and individual states and local areas have already put into place many programs to reduce ozone precursors. However, ships are significant contributors to ozone in many areas and states will need additional reductions in a timely manner to help them meet their air quality goals.

3.2.4 Projected Ozone Air Quality

Levels of ozone in the ambient air are expected to continue to be a problem into the future. Without further action, emissions from ships will contribute a larger share to the projected levels of ozone as emissions from other sources decrease. In this section we present information on projected levels of ozone in 2020, ships' contribution to these levels and the improvements which would occur with an ECA.

3.2.4.1 Projected Ozone Levels without an ECA

Figure 3.2-6 presents the projected seasonal average of daily 8-hour maximum ozone concentrations for the continental U.S. based on the inventory projections described in Section 2.4[L] Concentrations over most of the U.S. are in the 40 to 50 ppb range with a few scattered areas being lower, 30 to 40 ppb, or higher, up to > 70 ppb.

[K] The definition for standard temperature and pressure varies but both the U.S. EPA and the National Institute of Standards and Technology use 20° Celsius and an atmospheric pressure of 1013 mb.
[L] As discussed in Section 3.2.5.1.2 the air quality modeling domain only covers the continental United States.

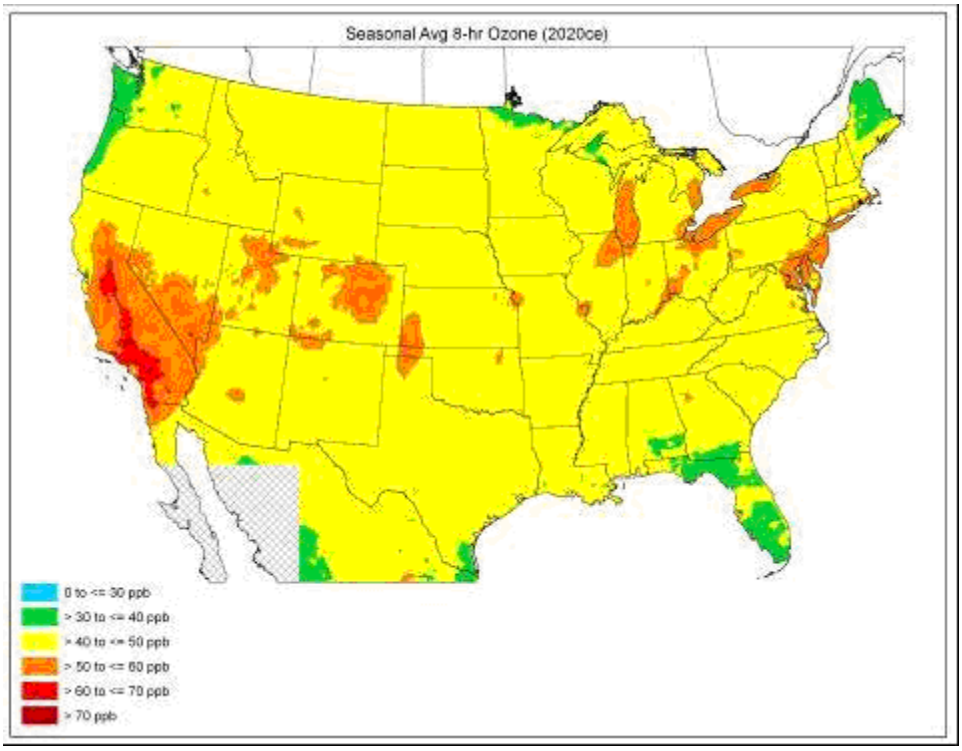

Figure 3.2-6 Seasonal Average of Daily 8-hour Maximum Ozone Concentrations in 2020 without an ECA

Even with the implementation of all current U.S. state and federal regulations, including the Acid Rain program and the NO_X SIP call which target SO_X and NO_X emissions that cause air quality issues far from power plants, nonroad and on-road diesel rules and the Tier II rule for highway vehicles, there are projected to be many areas in the U.S. with levels of ozone which are above health standards.[M] Emission reductions from the ECA designation would be helpful for U.S. states and counties in attaining and maintaining the ozone NAAQS and WHO AQG.

3.2.4.2 Contribution of Ships to Projected Ozone Levels

Emissions of NO_X from ships have a significant impact on ambient ozone concentrations. The contribution from ships were determined by comparing model results in two future year control runs, one with all sources and one without ships. Figure 3.2-7 illustrates the projected percentage contribution of ships to average daily maximum 8-hour ozone concentrations in 2020. The percentage contribution of ships to average daily maximum 8-hour ozone concentrations is projected to be between 5 and 15% throughout the gulf coast, the pacific coast and the southern east coast, with southern California experiencing contributions from ships of greater than 15%. The impacts of ship emissions on ozone concentrations would extend well inland, diminishing with distance from a coast. As can be

[M] See Chapter 5, Section 5.4 for more information about existing emission reduction programs to control land-based and other marine sources.

seen in Figure 3.2-7, the projected contribution of ships to ozone concentrations in many inland areas is up to 2%.

The absolute contribution of ships to 8-hour ozone concentrations is shown in Figure 3.2-8. This shows that the contribution from ships to 8-hour ozone concentrations is projected to be greater than 0.2 ppb for much of the country, while most coastal areas are projected to show impacts greater than 2.0 ppb.

The modeling projections clearly show that ships affect air quality on all the U.S. coastlines. This is to be expected since ships operate along all the U.S. coastlines. It can be concluded from looking at these results that emissions from ships need to be controlled in order to achieve ozone reductions, even in inland areas and areas without ports.

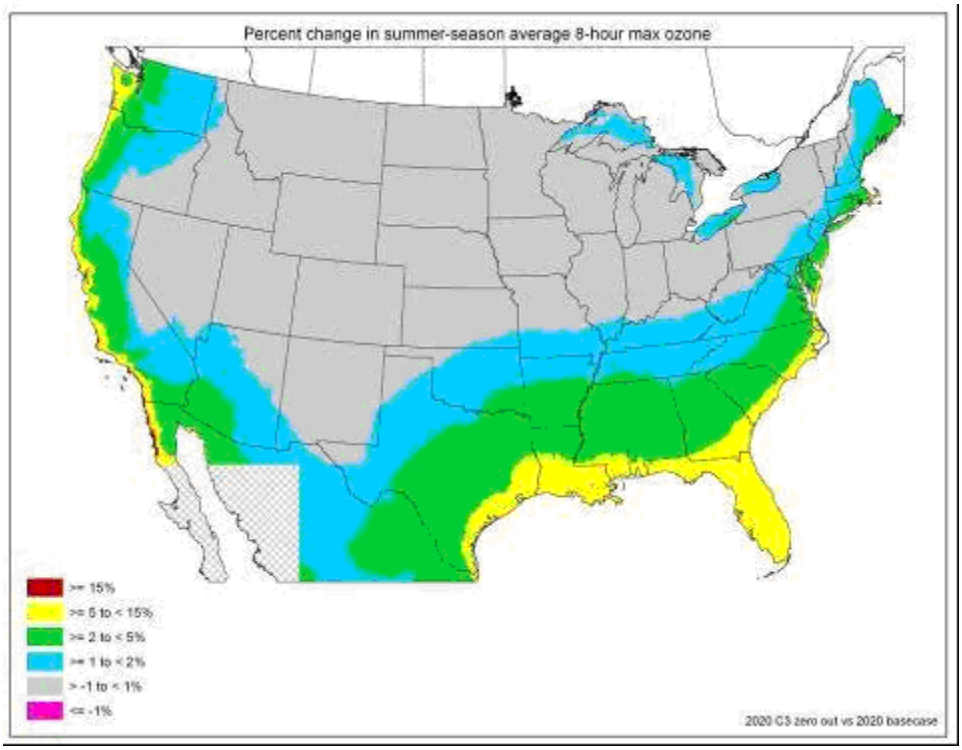

Figure 3.2-7 Percentage Contribution of Ships to Summertime Maximum 8-hour Average Ozone Concentrations in 2020

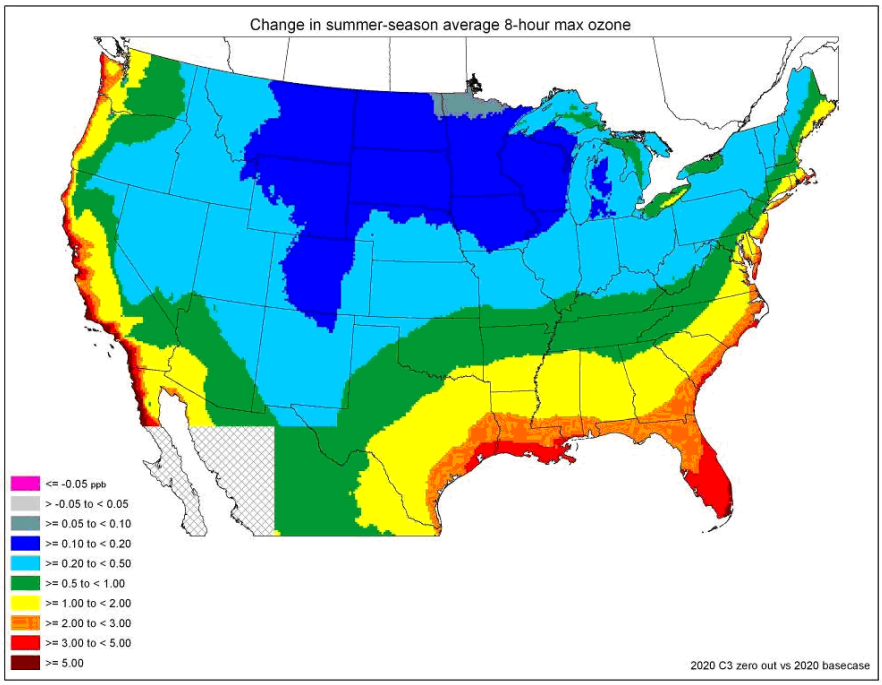

Figure 3.2-8 Absolute Contribution of Ships to Summertime Maximum 8-hour Average Ozone Concentrations in 2020

3.2.4.3 Projected Ozone Levels with an ECA

The impacts of the proposed ECA were determined by comparing the model results in the 2020 control run against the baseline simulation of the same year. According to air quality modeling performed for this analysis, the emission standards are expected to provide significant nationwide improvements in ozone levels.

Figures 3.2-9 and 3.2-10 present the projected percentage and absolute summertime maximum 8-hour average ozone improvements in 2020 if an ECA were enacted 200 nm from the U.S. shoreline. The ozone improvements are significant and extend inland including the states of Arizona, Missouri, Kentucky, Pennsylvania and New York. The entire U.S. coastline will experience improvements in their air quality from an ECA designation.

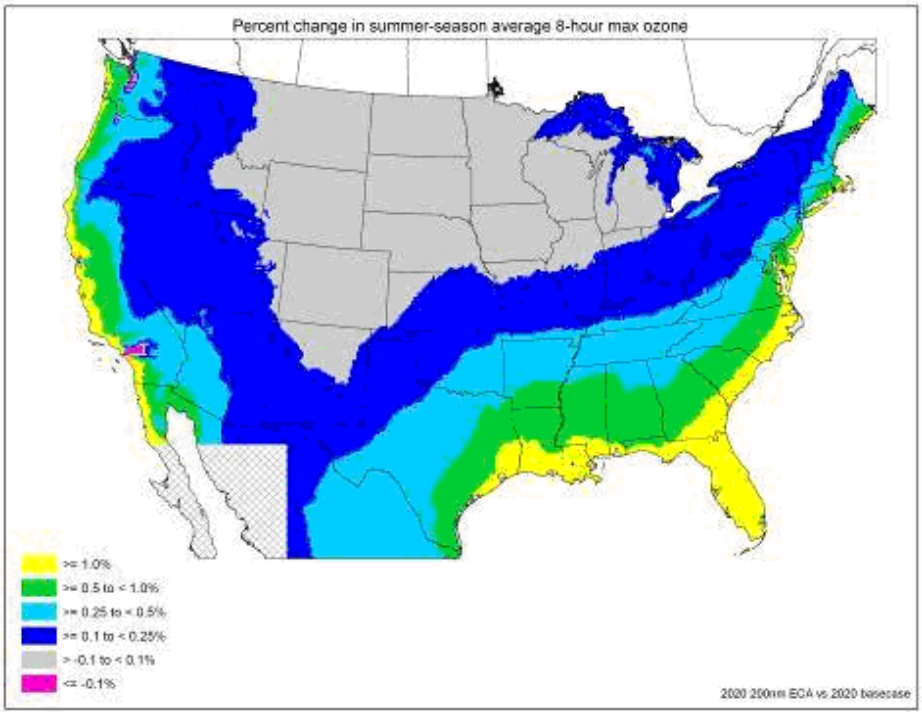

Figure 3.2-9 Percent Improvement in Summertime Maximum 8-hour Average Ozone Concentrations in 2020 Resulting from the Application of the Proposed ECA

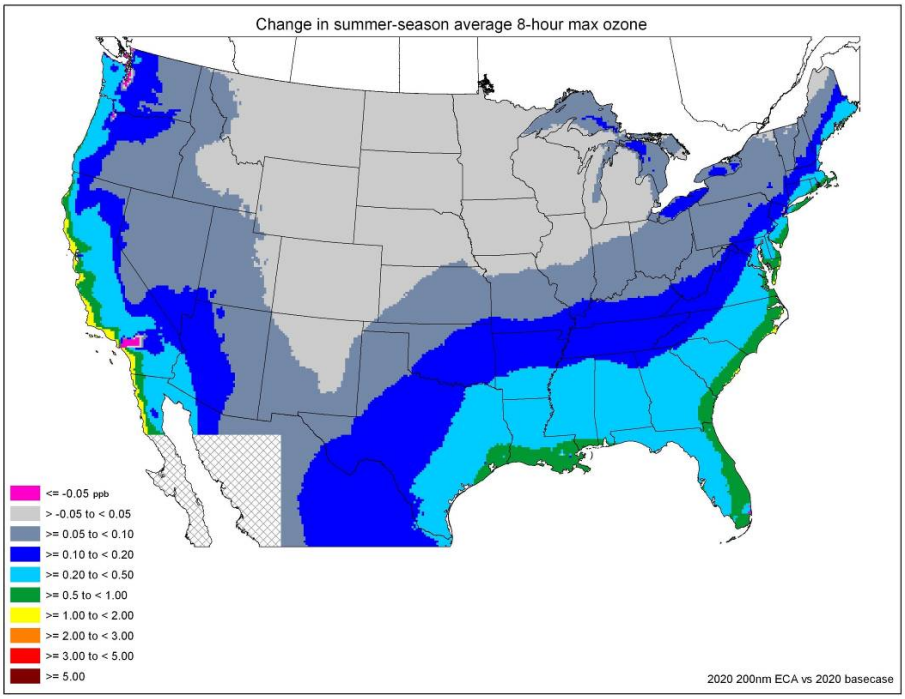

Figure 3.2-10 Absolute Improvement in Summertime Maximum 8-hour Average Ozone Concentrations in 2020 Resulting from the Application of the Proposed ECA

While the ECA designation would reduce ozone levels generally and provide national ozone-related health benefits, this is not always the case at the local level. The air quality modeling projects that in a few areas ozone levels will get higher because of the NO_X

disbenefit phenomenon. Due to the complex photochemistry of ozone production, NO_X emissions lead to both the formation and destruction of ozone, depending on the relative quantities of NO_X, VOC, and ozone formation catalysts such as the OH and HO_2 radicals. In areas dominated by fresh emissions of NO_X, ozone catalysts are removed via the production of nitric acid which slows the ozone formation rate. Because NO_X is generally depleted more rapidly than VOC, this effect is usually short-lived and the emitted NO_X can lead to ozone formation later and further downwind. The terms "NO_X disbenefits" or "ozone disbenefits" refer to the ozone increases that result when reducing Ox emissions in localized areas. According to the NARSTO Ozone Assessment, disbenefits are generally limited to small regions within specific urban cores and are surrounded by larger regions in which NO_X control is beneficial.[123] It is important to note the following as well: there is a level of NO_X control where enough NO_X will have been reduced to result in decreases in ambient ozone concentrations, this modeling does not include future VOC or NO_X controls that local areas are planning, and reductions in NO_X are not only important to help reduce ozone but also to help reduce $PM_{2.5}$.

3.2.5 Air Quality Modeling Methodology

When considering the potential effects of any particular air quality regulation, it is common practice to apply a photochemical air quality modeling system to estimate the change in air quality expected to occur with the emissions reductions proposed as part of the control program. At their root level, air quality models are quantitative approximations of the numerous complex physical and chemical interactions in the atmosphere that determine the formation and fate of air pollutants in the atmosphere. The U.S. government has traditionally used air quality modeling results to support policy decisions and as inputs into regulatory impact analyses. As part of this exercise, we have completed several air quality modeling simulations to look at the impact of a potential ECA application on future air pollution levels over the United States.

This section of the document describes the air quality modeling performed by the U.S. government in support of the ECA application. A fine-scale, national air quality modeling analysis was performed to estimate the effect in 2020 of the proposed ECA emissions reductions on future: 8-hour ozone concentrations, annual fine particulate matter ($PM_{2.5}$) concentrations, visibility levels, and acid deposition to watersheds and ecosystems. The following text will describe: the air quality model that was used, how it was applied, how the model inputs were developed, how the model was evaluated, and for what scenarios it was applied.

3.2.5.1 Modeling Methodology

For this analysis, we used a 2002-based, multi-pollutant modeling platform to assess the impacts of reduced marine emissions from the application of an ECA. This platform represents a structured system of connected modeling-related tools and data that provide a consistent and transparent basis for assessing the air quality response to changes in emissions, meteorology, and/or model formulation. The base year of data used to construct this platform includes emissions and meteorology for 2002. The platform was developed by the U.S. EPA's Office of Air Quality Planning and Standards in collaboration with the Office of

Research and Development and is intended to support a variety of regulatory and research model applications and analyses.

There are four key elements to the modeling platform, all of which will be described in more detail in subsequent sections. The key elements are:

- the selected air quality model;
- the emissions, meteorological, and initial and boundary concentration data which are input to the model;
- the emissions and meteorological models (or pre-processors) used to prepare the input data in the form and format needed for air quality model simulations; and
- the predicted concentration and deposition values predicted by the model.

3.2.5.1.1 Air Quality Model

The CMAQ modeling system is a non-proprietary comprehensive three-dimensional, grid-based Eulerian air quality model designed to estimate the formation and fate of oxidant precursors, primary and secondary PM concentrations and deposition, over regional and urban spatial scales for given input sets of meteorological conditions and emissions.[124,125,126] CMAQ is a publicly available, peer reviewed[N], state-of-the-science model consisting of a number of science attributes that are critical for simulating the oxidant precursors and non-linear organic and inorganic chemical relationships associated with the formation of sulfate, nitrate, and organic aerosols. CMAQ also simulates the transport and removal of directly emitted particles which are speciated as elemental carbon, crustal material, nitrate, sulfate, and organic aerosols. The CMAQ model version 4.6 was most recently peer-reviewed in February of 2007 for the U.S. EPA as reported in the "Third Peer Review of the CMAQ Model."[127] The CMAQ model is a well-known and well-respected tool and has been used in numerous national and international applications.[128,129,130]

The CMAQ modeling system is designed as an "open system" where new scientific algorithms and mechanisms can be utilized and evaluated in conjunction with CMAQ processes. Model parameterizations may also be modified to test performance characteristics of dynamical-chemical processes within model simulations, such as tropospheric ozone, visibility, acid deposition, and particulate matter. CMAQ offers a multi-pollutant (i.e., ozone, particulates, acid deposition, and nitrogen loading) capability via a generalized chemistry mechanism, general numerical solver, and comprehensive description of gaseous and aqueous chemistry and modal aerosol dynamics. CMAQ was also designed with scaleable atmospheric dynamics and generalized coordinates to address multi-scale capabilities (e.g. regional or local scale) depending on a user-defined model resolution. To resolve atmospheric dynamics at local scales, CMAQ utilizes a set of governing equations for compressible non-hydrostatic atmospheres expressed in a generalized coordinate system. The

[N] Community Modeling & Analysis System (CMAS) – Reports from the CMAQ Review Process can be found at: http://www.cmascenter.org/r_and_d/cmaq_review_process.cfm?temp_id=99999 .

generalized coordinate system allows various vertical coordinates and map projections to be used and resolves the necessary grid and coordinates transformations.

This 2002 multi-pollutant modeling platform used the latest publicly-released CMAQ version 4.6[O] with a few minor changes and new features made internally by the U.S. EPA CMAQ model developers, all of which reflects updates to earlier versions in a number of areas to improve the underlying science. The model enhancements in CMAQ v4.6.1 include:

1) an in-cloud sulfate chemistry module that accounts for the nonlinear sensitivity of sulfate formation to varying pH;

2) an improved vertical asymmetric convective mixing module (ACM2) that allows in-cloud transport from a source layer to all other-in cloud layers (combined non-local and local closure scheme);

3) a heterogeneous reaction involving nitrate formation (gas-phase reactions involving N_2O_5 and H_2O);

4) the heterogeneous N_2O_5 reaction probability is now temperature- and humidity-dependent,

5) an updated version of the ISORROPIA aerosol thermodynamics module including improved representation of aerosol liquid water content and correction in activity coefficients for temperature other than 298K, and

6) an updated gas-phase chemistry mechanism, Carbon Bond 05 (CB05) and associated Euler Backward Iterative (EBI) solver, with extensions to model explicit concentrations of air toxic species.

3.2.5.1.2 Air Quality Model Domain and Configuration

The CMAQ modeling analyses were performed for three separate domains, as shown in Figure 3.2-11. This modeling used a parent horizontal grid of 36 km with two nested, finer-scale 12 km grids covering the Eastern and Western U.S. (i.e., EUS and WUS grids respectively).[P,Q] The model extends vertically from the surface to 100 millibars using a sigma-pressure coordinate system. Air quality conditions at the outer boundary of the 36 km domain were taken from the global GEOS-Chem model and did not change over the simulated scenarios. The 36 km grid was only used to establish the incoming air quality concentrations along the boundaries of the 12 km grids. All of the modeling results assessing the air quality impacts of emissions reductions from the application of ECA controls were

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[O] CMAQ version 4.6 was released on September 30, 2006. It is available from the Community Modeling and Analysis System (CMAS) as well as previous peer-review reports at: http://www.cmascenter.org.

[P] We were unable to consider effects beyond the 48-State area due to the unavailability of gridded meteorological data for locations like Alaska and Hawaii.

[Q] In the overlapping portion of the two fine grids we used the WUS results for the States of MT, WY, CO, and NM, and the EUS results for ND, MN, SD, IA, NE, MO, KS, OK, and TX.

taken from the 12 km grids. Table 3.2-1 provides some basic geographic information regarding the CMAQ domains. Table 3.2-2 provides information on the vertical structure of the CMAQ modeling as well as the model which provided meteorological inputs. Table 3.2-3 indicates which CMAQ configuration options were chosen for this analysis.

Figure 3.2-11. Map of the CMAQ Modeling Domains. The black outer box denotes the 36 km national modeling domain; the red inner box is the 12 km western U.S. fine grid; and the blue inner box is the 12 km eastern U.S. fine grid.

Table 3.2-1. Geographic Elements of Domains used in the ECA Modeling.

CMAQ MODELING CONFIGURATION			
	National Grid	**Western U.S. Fine Grid**	**Eastern U.S. Fine Grid**
Map Projection	Lambert Conformal Projection		
Grid Resolution	36 km	12 km	12 km
Coordinate Center	97 deg W, 40 deg N		
True Latitudes	33 deg N and 45 deg N		
Dimensions	148 x 112 x 14	213 x 192 x 14	279 x 240 x 14
Vertical extent	14 Layers: Surface to 100 millibar level (see Table 3-XX)		

Table 3.2-2. Vertical Layer Structure for MM5 and CMAQ (heights are layer top).

CMAQ LAYERS	MM5 LAYERS	SIGMA P	APPROXIMATE HEIGHT (M)	APPROXIMATE PRESSURE (MB)
0	0	1.000	0	1000
1	1	0.995	38	995
2	2	0.990	77	991
3	3	0.985	115	987
	4	0.980	154	982
4	5	0.970	232	973
	6	0.960	310	964
5	7	0.950	389	955
	8	0.940	469	946
6	9	0.930	550	937
	10	0.920	631	928
	11	0.910	712	919
7	12	0.900	794	910
	13	0.880	961	892
	14	0.860	1,130	874
8	15	0.840	1,303	856
	16	0.820	1,478	838
	17	0.800	1,657	820
9	18	0.770	1,930	793
	19	0.740	2,212	766
10	20	0.700	2,600	730
	21	0.650	3,108	685
11	22	0.600	3,644	640
	23	0.550	4,212	595
12	24	0.500	4,816	550
	25	0.450	5,461	505
	26	0.400	6,153	460
13	27	0.350	6,903	415
	28	0.300	7,720	370
	29	0.250	8,621	325
	30	0.200	9,625	280
14	31	0.150	10,764	235
	32	0.100	12,085	190
	33	0.050	13,670	145
	34	0.000	15,674	100

Table 3.2-3. Additional Details Regarding the CMAQ Model Configuration.

GAS-PHASE CHEMICAL MECHANISMKRER	CB05
Gas-Phase Chemical Solver	Euler Backward Iterative (EBI) scheme
PM Module	AERO4 aerosol module which contains mechanisms dealing with sea salt emissions. Three-mode approach: One coarse mode, two fine modes with variable standard deviations.
Inorganic PM module	ISORROPIA

Organic PM module	Updated SOA module based on Odum/Griffin et al., (1997, 1999)
Advection Scheme (vertical and horizontal)	Piecewise Parabolic Method (PPM)
Planetary Boundary Layer Scheme	Asymmetric Convective Mixing module (ACM2) scheme which permits gradual layer-by-layer downward mixing through compensatory subsidence
Dry Deposition	M3DRY module modified RADM scheme
Aqueous Chemistry	RADM Bulk scheme
Cloud Scheme	RADM Cloud scheme
Vertical Coordinate	Terrain-following Sigma coordinate

The 36 km and both 12 km CMAQ modeling domains were modeled for the entire year of 2002. We also modeled ten days at the end of December 2001 as a model "ramp up" period. These days are used to minimize the effects of initial conditions and are not considered as part of the output analyses. All 365 model days were used in the calculations of the ECA impacts on annual average levels of $PM_{2.5}$. For the 8-hour ozone results, we only used the modeling results from the period between May 1 and September 30, 2002. This 153-day period generally conforms to the ozone season across most parts of the U.S. and contains the majority of days with observed high ozone concentrations in 2002.

3.2.5.1.3 Air Quality Model Inputs

The key inputs to the CMAQ model include emissions from anthropogenic and biogenic sources, meteorological data describing atmospheric states and motions, and initial and boundary conditions. A summary of these three modeling components are described below.

3.2.5.1.3.1 Emissions Inventory Data Inputs

With the exception of the marine emissions discussed in Section 2 of this document, the CMAQ gridded 2002 emissions input data were based on emissions from the 2002 National Emissions Inventory (NEI) version 3.0. This inventory includes emissions of criteria pollutants[R] from point, stationary area, and mobile source categories. With the exception of California[S], monthly onroad and nonroad emissions were generated from the National Mobile Inventory Model (NMIM) using versions of MOBILE6.0 and NONROAD2005 consistent with recent national rule analyses.[T,U] The 2002-based platform and its associated chemical

[R] Criteria pollutant emissions include sulfur dioxide, oxides of nitrogen, carbon monoxide, volatile organic compounds, ammonia, and fine particles.

[S] The California Air Resources Board submitted annual emissions for California. These were allocated to monthly resolution prior to emissions modeling using data from the National Mobile Inventory Model (NMIM).

[T] MOBILE6 version was used in the Mobile Source Air Toxics Rule: *Regulatory Impact Analysis for Final Rule:* Control *of Hazardous Air Pollutants from Mobile Sources*, U.S. Environmental Protection Agency, Office of

mechanism (CB05) employs updated speciation profiles using data included in the SPECIATE4.0 database.[V] The 2002-based platform also incorporates several temporal profile updates for both mobile and stationary sources.

The 2002-based platform includes emissions for a 2002 base year model evaluation case, a 2002 base case and a 2020 future base case. The model evaluation case uses prescribed burning and wildfire emissions specific to 2002, which were developed and modeled as day-specific, location-specific emissions using an updated version of Sparse Matrix Operator Kernel Emissions (SMOKE) system, version 2.3, which computes plume rise and vertically allocates the fire emissions. SMOKE also provides mobile, area, and point source emissions as gridded, temporalized, and speciated data inputs to CMAQ (Houyoux and Vukovich, 1999).[131] The 2002 evaluation case also includes continuous emissions monitoring (CEM) data for 2002 for electric generating units (EGUs) with CEMs. The 2002 and projection year baselines include multi-year averages for the fire sector and EGU emissions that are temporally allocated based on a combination of multi-year average and 2002 temporal profiles. Projections from 2002 were developed to account for the expected impact of national regulations, consent decrees or settlements, known plant closures, and, for some sectors, activity growth. Biogenic emissions were processed using the Biogenic Emissions Inventory System (BEIS) version 3.13.

3.2.5.1.3.2 Meteorological Data Inputs

The CMAQ gridded meteorological input data for the entire year of 2002 were derived from simulations of the Pennsylvania State University / National Center for Atmospheric Research Mesoscale Model. This model, commonly referred to as MM5, is a limited-area, nonhydrostatic, terrain-following system that solves for the full set of physical and thermodynamic equations which govern atmospheric motions.[132] Meteorological model input fields were prepared separately for each of the domains shown in Figure 3.2-11 above. The 36 km national domain was modeled using MM5 v.3.6.0 and the 12 km Eastern U.S grid was modeled with MM5 v3.7.2. Both of these two sets of meteorological inputs were developed by the U.S. EPA. For the 12 km western U.S. grid, we utilized existing MM5 meteorological model data prepared by the Western Regional Air Partnership.[133] All three sets of MM5 model runs were conducted in 5.5 day segments with 12 hours of overlap for spin-up purposes. Additionally, all three domains contained 34 vertical layers with an approximately 38 m deep surface layer and a 100 millibar top. The MM5 and CMAQ vertical structures are shown in Table 3.2-2 and do not vary by horizontal grid resolution.

Transportation and Air Quality, Assessment and Standards Division, Ann Arbor, MI 48105, EPA420-R-07-002, February 2007.

[U] NONROAD2005 version was used in the proposed rule for small spark ignition (SI) and marine SI rule: Draft Regulatory Impact Analysis: *Control of Emissions from Marine SI and Small SI Engines, Vessels, and Equipment* , U.S. Environmental Protection Agency, Office of Transportation and Air Quality, Office of Transportation and Air Quality, Assessment and Standards Division, Ann Arbor, MI, EPA420-D-07-004, April 2007.

[V] See http://www.epa.gov/ttn/chief/software/speciate/index.html for more details.

The meteorological outputs from MM5 were processed to create model-ready inputs for CMAQ using the Meteorology-Chemistry Interface Processor (MCIP) version 3.1 to derive the specific inputs to CMAQ, for example: horizontal wind components (i.e., speed and direction), temperature, moisture, vertical diffusion rates, and rainfall rates for each grid cell in each vertical layer. Before initiating the air quality simulations, an evaluation was conducted to identify the biases and errors associated with the meteorological modeling inputs. The U.S. EPA 2002 MM5 model performance evaluations used an approach which included a combination of qualitative and quantitative analyses to assess the adequacy of the MM5 simulated fields. More detail on the meteorological modeling evaluations can be found in the following references.[134,135] The general conclusion of each of these meteorological evaluations was that the simulated meteorology reproduced the actual meteorology with sufficient accuracy for them to be used in subsequent air quality analyses.

3.2.5.1.3.3 Initial and Boundary Conditions Data Inputs

The lateral boundary and initial species concentrations are provided by a three-dimensional global atmospheric chemistry model, the GEOS-CHEM model.[136] The global GEOS-CHEM model simulates atmospheric chemical and physical processes driven by assimilated meteorological observations from the NASA's Goddard Earth Observing System (GEOS). This model was run for 2002 with a grid resolution of 2.0 degree x 2.5 degree (latitude-longitude) and 20 vertical layers. The predictions were used to provide one-way dynamic boundary conditions at three-hour intervals and an initial concentration field for the 36 km CMAQ simulations. The 36 km coarse grid modeling was used as the initial/boundary conditions for the 12 km EUS and WUS finer grid modeling. More information is available about the GEOS-CHEM model and other applications using this tool at: http://www-as.harvard.edu/chemistry/trop/geos.

3.2.5.1.4 Air Quality Model Evaluation

An operational model performance evaluation for ozone and $PM_{2.5}$ and its related speciated components was conducted using 2002 State/local monitoring data in order to estimate the ability of the CMAQ modeling system to replicate the base year concentrations for the 12-km EUS and WUS grids. This evaluation principally comprises statistical assessments of model versus observed pairs that were paired in space and time on a daily or weekly basis, depending on the sampling frequency of each monitoring network. For any time periods with missing ozone and $PM_{2.5}$ observations we excluded the CMAQ predictions from those time periods in our calculations. It should be noted when pairing model and observed data that each CMAQ concentration represents a grid-cell volume-averaged value, while the ambient network measurements are made at specific locations. In conjunction with the model performance statistics, we also provide spatial plots for individual monitors of the calculated bias and error statistics (defined below). Statistics were generated for the 12-km

EUS and WUS grids and five large subregions.[W] The Atmospheric Model Evaluation Tool (AMET) was used to conduct the evaluation described in this document.[137]

The ozone evaluation primarily focused on observed hourly ozone concentrations and eight-hour daily maximum ozone concentrations above a threshold of 40 ppb. The ozone model performance evaluation was limited to the ozone season modeled for the ECA: May, June, July, August, and September. Ozone ambient measurements for 2002 were obtained from the Air Quality System (AQS) Aerometric Information Retrieval System (AIRS). A total of 1178 ozone measurement sites were included for evaluation. The ozone data were measured and reported on an hourly basis.

The $PM_{2.5}$ evaluation focuses on $PM_{2.5}$ total mass and its components including sulfate (SO_4), nitrate (NO_3), total nitrate (TNO_3=NO_3+HNO_3), ammonium (NH_4), elemental carbon (EC), and organic carbon (OC). The $PM_{2.5}$ performance statistics were calculated for each month and season individually and for the entire year, as a whole. Seasons were defined as: winter (December-January-February), spring (March-April-May), summer (June-July-August), and fall (September-October-November). $PM_{2.5}$ ambient measurements for 2002 were obtained from the following networks for model evaluation: Speciation Trends Network (STN, total of 199 sites), Interagency Monitoring of Protected Visual Environments (IMPROVE, total of 150), and Clean Air Status and Trends Network (CASTNet, total of 83). The pollutant species included in the evaluation for each network are listed in Table 3.2-4. For $PM_{2.5}$ species that are measured by more than one network, we calculated separate sets of statistics for each network.

Table 3.2-4. $PM_{2.5}$ Monitoring Networks and Pollutants Species Included in the CMAQ Performance Evaluation.

AMBIENT MONITORING NETWORKS	PARTICULATE SPECIES						
	$PM_{2.5}$ Mass	SO_4	NO_3	TNO_3	NH_4	EC	OC
IMPROVE	X	X	X		X	X	X
CASTNet		X		X	X		
STN	X	X	X		X	X	X
Note that $TNO_3 = (NO_3 + HNO_3)$							

There are various statistical metrics available and used by the science community for model performance evaluation. The four evaluation statistics used to evaluate CMAQ performance were two bias metrics, normalized mean bias and fractional bias; and two error metrics, normalized mean error and fractional error.

[W] The subregions are defined by States where: Midwest is IL, IN, MI, OH, and WI; Northeast is CT, DE, MA, MD, ME, NH, NJ, NY, PA, RI, and VT; Southeast is AL, FL, GA, KY, MS, NC, SC, TN, VA, and WV; Central is AR, IA, KS, LA, MN, MO, NE, OK, and TX; West is AK, CA, OR, WA, AZ, NM, CO, UT, WY, SD, ND, MT, ID, and NV.

The "acceptability" of model performance was judged by comparing our CMAQ 2002 performance results to the range of performance found in recent regional ozone and $PM_{2.5}$ model applications. These other modeling studies represent a wide range of modeling analyses which cover various models, model configurations, domains, years and/or episodes, chemical mechanisms, and aerosol modules. Overall, the statistical calculations of model bias and error indicate that the CMAQ predicted ozone and $PM_{2.5}$ concentrations for 2002 are within the range or close to that found in recent U.S. EPA applications.[138] Figures 3.2-12 to 3.2-15 show the seasonal aggregate normalized mean bias for 8-hourly ozone and $PM_{2.5}$ over the two 12-km grids. The CMAQ model performance results give us confidence that our applications of CMAQ using this 2002 modeling platform provide a scientifically credible approach for the impacts of ECA controls on ozone and $PM_{2.5}$ concentrations, visibility levels, and acid deposition amounts.

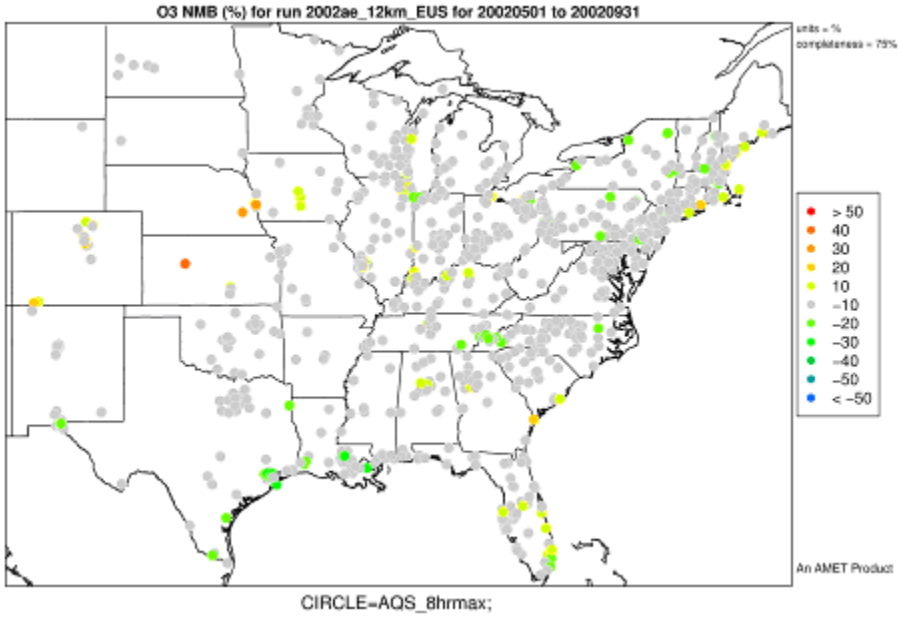


Figure 3.2-12. Normalized Mean Bias (%) of hourly ozone (40 ppb threshold) by monitor for 12-km Eastern U.S. domain, seasonal aggregate

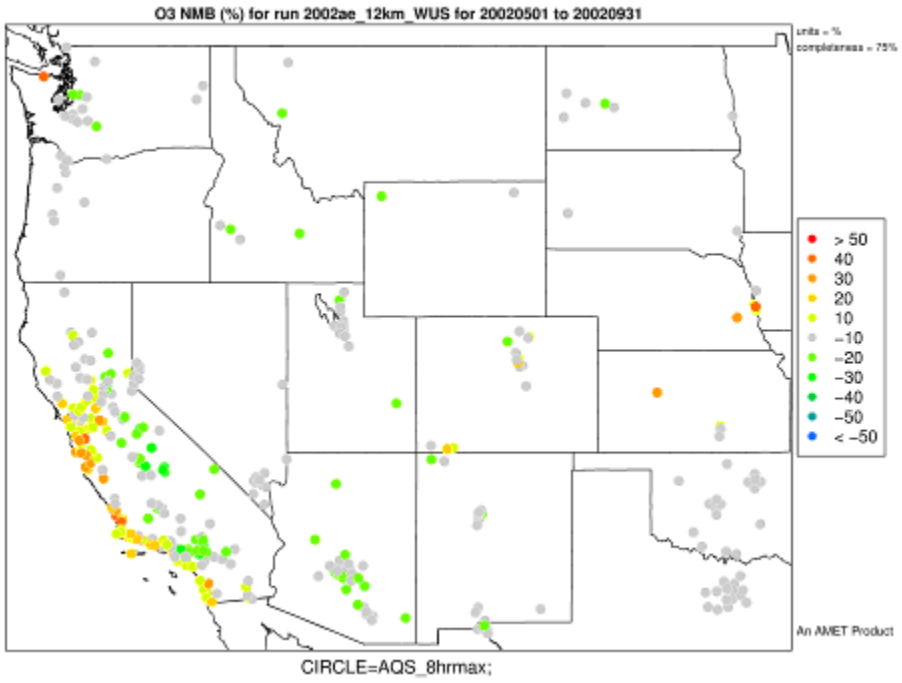


Figure 3.2-13. Normalized Mean Bias (%) of hourly ozone (40 ppb threshold) by monitor for 12-km Western U.S. domain, seasonal aggregate.

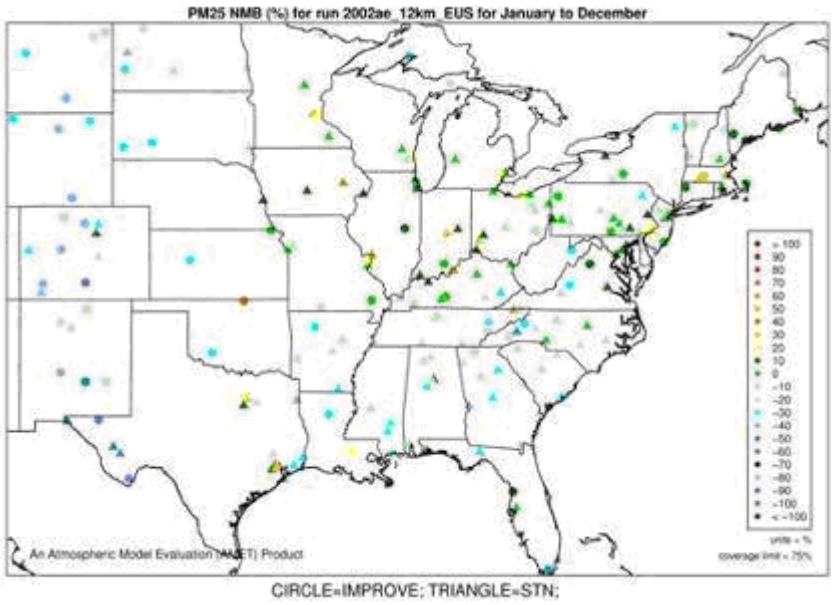


Figure 3.2-14. Normalized Mean Bias (%) of annual $PM_{2.5}$ by monitor for 12-km Eastern U.S. domain, 2002

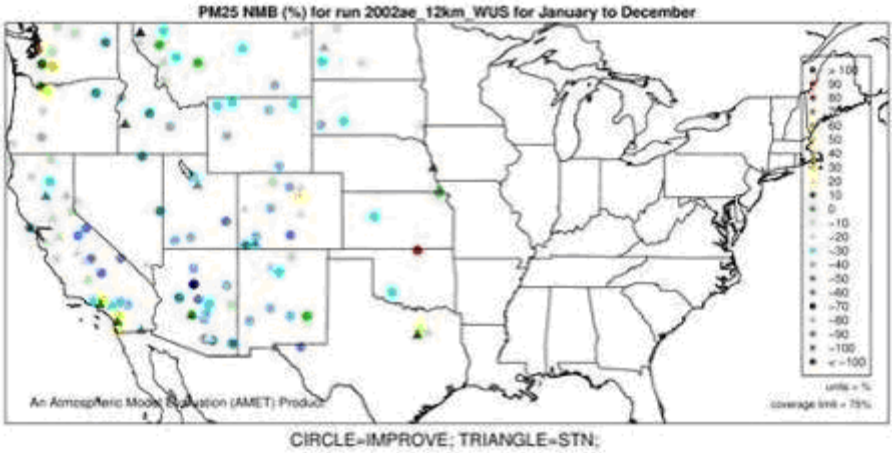


Figure 3.2-15. Normalized Mean Bias (%) of annual PM2.5 by monitor for 12-km Western U.S. domain, 2002

3.3 Impacts on Ecosystems

3.3.1 Sulfur and Nitrogen Deposition (overview)

Large ships release emissions over a wide area, and depending on prevailing winds and other meteorological conditions, these emissions may be transported hundreds and even thousands of kilometers across North America. Sections 3.1 and 3.2 discuss the results of U.S. air quality modeling which documents this phenomenon. Overall these engines emit a large amount of NO_X, SO_X and direct PM which impact not only ambient air concentrations but also contribute to deposition of nitrogen and sulfur in many sensitive ecological areas throughout the U.S.

Sulfur in marine fuel is primarily emitted as SO_2, with a small fraction (about two percent) being converted to SO_3.[139,140,141] SO_3 almost immediately forms sulfate and is emitted as primary PM by the engine and consists of carbonaceous material, sulfuric acid, and ash (trace metals). Ships operating on high sulfur fuel therefore, emit large amounts of both SO_2 and sulfate PM. The vast majority of the primary PM is less than or equal to 2.5 μm in diameter, and accounts for the majority of the number of particles in exhaust, but only a small fraction of the mass of diesel PM. These particles also react in the atmosphere to form secondary PM, which exist there as a carbon core with a coating of organic carbon compounds, nitrate particles, or as sulfuric acid and ash, sulfuric acid aerosols, or sulfate particles associated with organic carbon.

At the same time, ships emit large amounts of NO and NO_2 (NO_X) emissions which are carried into the atmosphere where they may be chemically altered and transformed into new compounds. For example, NO_2 can also be further oxidized to nitric acid ($HNO_{3)}$ and can contribute in that form to the acidity of clouds, fog, and rain water and can also form ambient particulate nitrate (pNO_3) which may be deposited either directly onto terrestrial and aquatic ecosystems (“direct deposition”) or deposited onto land surfaces where it subsequently runs off and is transferred into downstream waters (“indirect deposition”).

Deposition of nitrogen and sulfur resulting from ship operations can occur either in a wet or dry form. Wet deposition includes rain, snow, sleet, hail, clouds, or fog. Dry deposition includes gases, dust, and minute particulate matters. Wet and dry atmospheric deposition of $PM_{2.5}$ delivers a complex mixture of metals (such as mercury, zinc, lead, nickel, arsenic, aluminum, and cadmium), organic compounds (such as polycyclic organic matter, dioxins, and furans) and inorganic compounds (such as nitrate, sulfate). Together these emissions from ships are deposited onto terrestrial and aquatic ecosystems across the U.S. contributing to the problems of ecosystem acidification, ecosystem nutrient enrichment, and ecosystem eutrophication.

Deposition of nitrogen and sulfur causes acidification, which alters biogeochemistry and affects animal and plant life in terrestrial and aquatic ecosystems across the U.S. Major effects include a decline in some forest tree species, such as red spruce and sugar maple; and a loss of biodiversity of fishes, zooplankton, and macro invertebrates. The sensitivity of terrestrial and aquatic ecosystems to acidification from nitrogen and sulfur deposition is predominantly governed by the earth's geology.

Biological effects of acidification in terrestrial ecosystems are generally linked to aluminum toxicity and decreased ability of plant roots to take up base cations. Decreases in the acid neutralizing capacity and increases in inorganic aluminum concentration contribute to declines in zooplankton, macro invertebrates, and fish species richness in aquatic ecosystems. Across the U.S., ecosystems will continue to be acidified by current NO_X and SO_X emissions from stationary sources, area sources, and mobile sources. For example, in the Adirondacks Mountains of New York State, the current rates of nitrogen and sulfur deposition exceed the amount that would allow recovery of the most acid sensitive lakes to a sustainable acid neutralizing capacity (ANC) level.

In addition to the role nitrogen deposition plays in acidification, it also causes ecosystem nutrient enrichment and eutrophication that alters biogeochemical cycles and harms animal and plant life such as native lichens and alters biodiversity of terrestrial ecosystems, such as forests and grasslands. Nitrogen deposition contributes to eutrophication of estuaries and coastal waters which result in toxic algal blooms and fish kills. For example, the Chesapeake Bay Estuary is highly eutrophic and 21 -30% of total nitrogen load comes from deposition. Freshwater ecosystems may also be impacted by nitrogen deposition, for example, high elevation freshwater lakes in the western U.S. experience adverse ecosystem changes at nitrogen deposition rates as low as 2 kg N/ha/yr.[142]

There are a number of important quantified relationships between nitrogen deposition levels and ecological effects. Certain lichen species are the most sensitive terrestrial taxa to nitrogen with species losses occurring at just 3 kg N/ha/yr in the Pacific Northwest of the U.S. and the southern portion of the State of California (See Figure 3-5 for the geographic distribution of these lichens in the continental U.S.). The onset of declining biodiversity was found to occur at levels of 5 kg N/ha/yr and above within grasslands in Minnesota and in Europe. Altered species composition of Alpine ecosystems and forest encroachment into temperate grasslands was found at 10 kg N/ha/yr and above in the U.S.

The biogeochemical cycle of mercury, a well-known neurotoxin, is closely tied to the sulfur cycle. Mercury is taken up by living organisms in the methylated form, which is easily bioaccumulated in the food web. Sulfate-reducing bacteria in wetland and lake sediments play a key role in mercury methylation. Changes in sulfate deposition have resulted in changes in both the rate of mercury methylation and the corresponding mercury concentrations in fish. In 2006, 3,080 fish advisories were issued in the U.S. due to the presence of methyl mercury in fish. Although sulfur deposition is important to mercury methylation, several other interrelated factors seem to also be related to mercury uptake, including low lake water pH, dissolved organic carbon, suspended particulate matter concentrations in the water column, temperature, and dissolved oxygen. In addition, the proportion of upland to wetland land area within a watershed, as well as wetland type and annual water yield, appear to be important.

3.3.1.1 Recent U.S. Deposition Data

Over the past two decades the U.S. has undertaken numerous efforts to reduce nitrogen and sulfur deposition across the U.S. Analyses of long-term monitoring data for the U.S. show that deposition of both nitrogen and sulfur compounds has decreased over the last 17 years although many areas continue to be negatively impacted by deposition. Deposition of inorganic nitrogen and sulfur species routinely measured in the U.S. between 2004 and 2006 were as high as 9.6 kg N/ha/yr and 21.3 kg S/ha/yr. Figures 3.3-1 and 3.3-2 show that annual total deposition (the sum of wet and dry deposition) decreased between 1989-1999 and 2004-2006 due to sulfur and NO_X controls on power plants, motor vehicles and fuels in the U.S. The data shows that reductions were more substantial for sulfur compounds than for nitrogen compounds. These numbers are generated by the U.S. national monitoring network and they likely underestimate nitrogen deposition because NH_3 is not measured. In the eastern U.S., where data are most abundant, total sulfur deposition decreased by about 36 percent between 1990 and 2005 while total nitrogen deposition decreased by 19 percent over the same time frame.[143]

The U.S. is concerned that both current ship emissions and projected future ship emissions will seriously erode environmental improvements that have been achieved in these ecologically sensitive areas. As the air quality modeling results in section 3.3.1.7 show, both nitrogen and sulfur deposition resulting from ship emissions impact a significant portion of ecologically sensitive areas in the U.S.

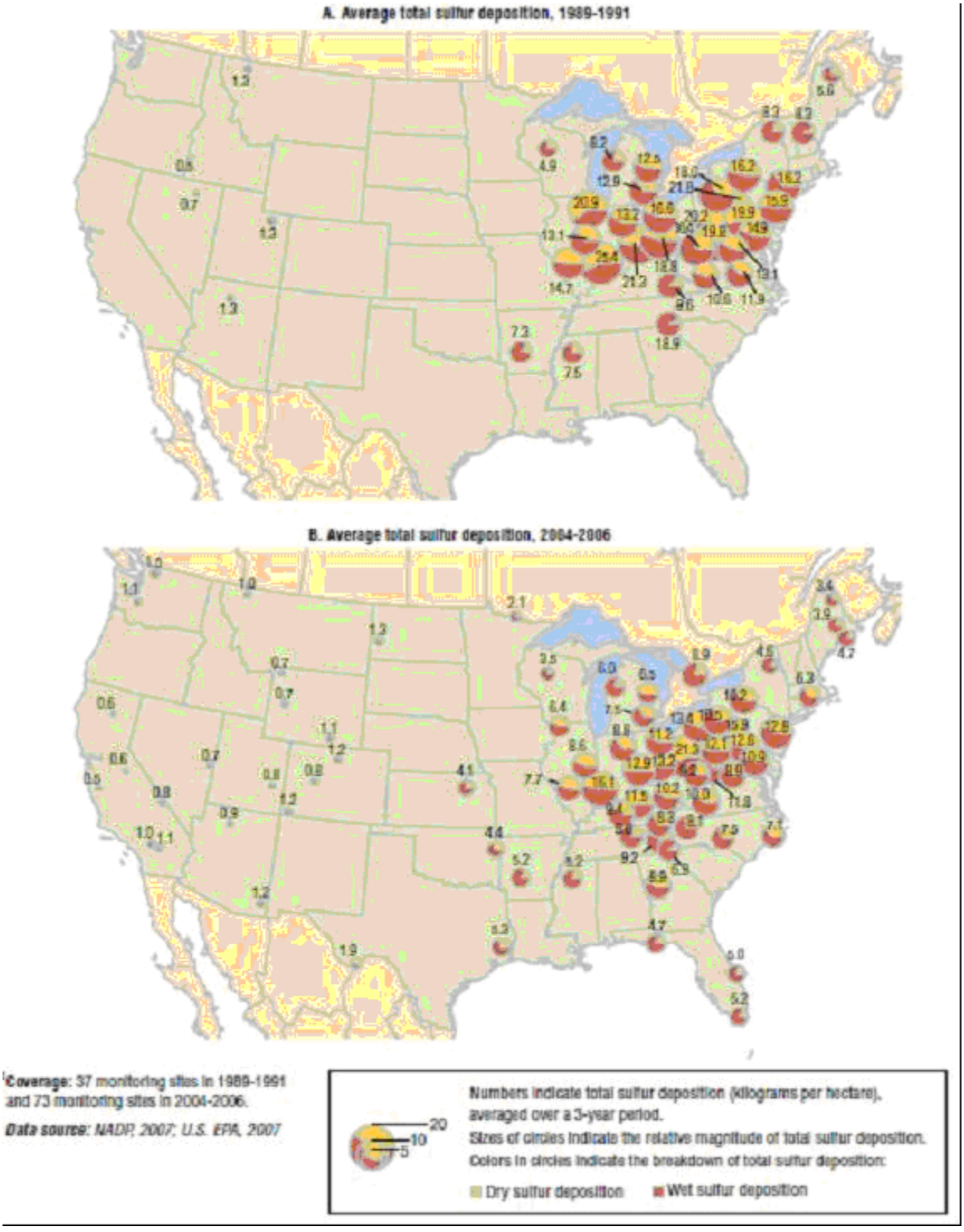


Figure 3.3-1 Total Sulfur Deposition in the Contiguous U.S., 1989-1991 and 2004 -2006

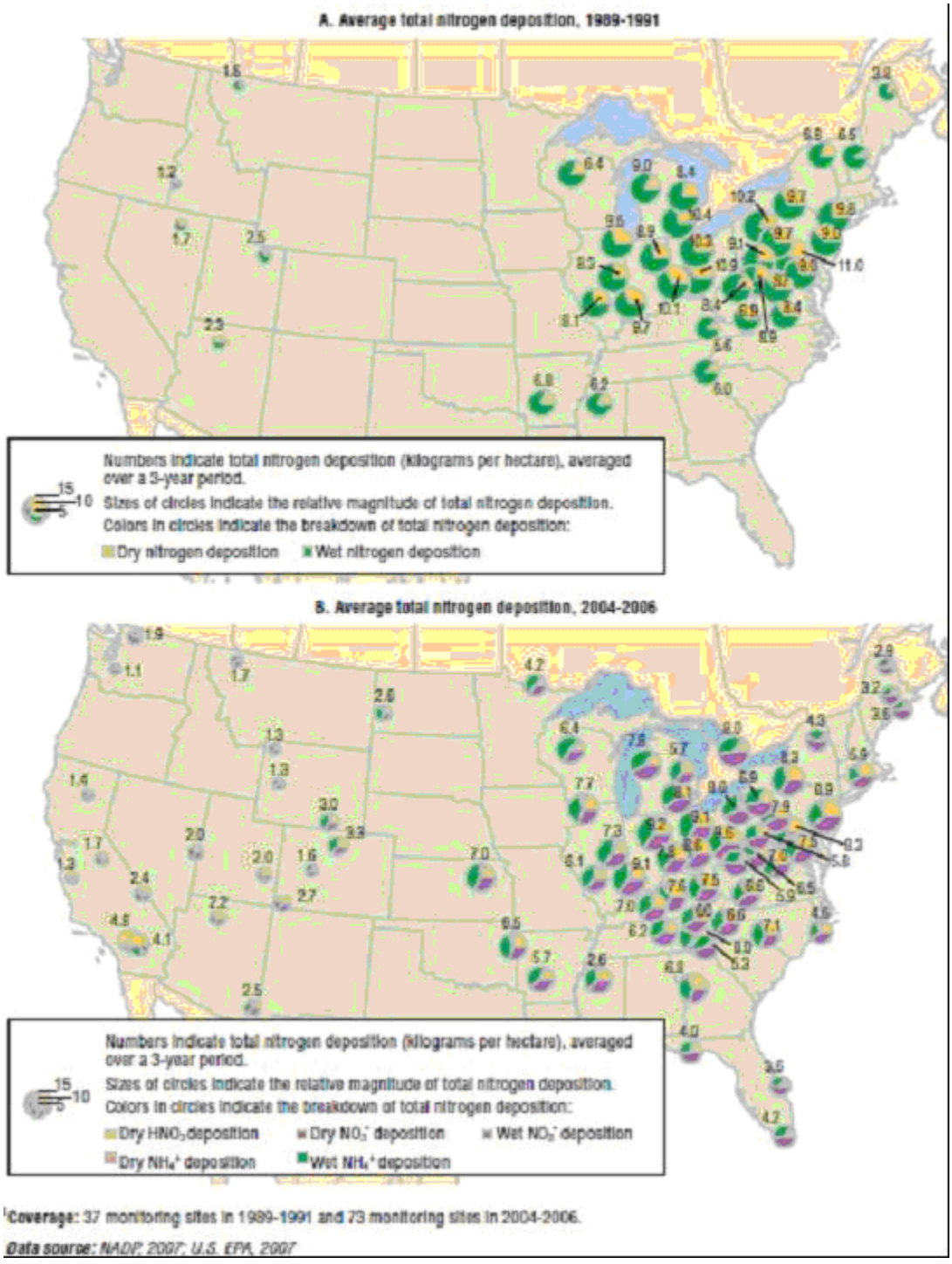


Figure 3.3-2 Total Nitrogen Deposition in the Contiguous U.S., 1989-1991 and 2004-2006

3.3.1.2 Areas Potentially Sensitive to Nitrogen and Sulfur Deposition in the U.S.

Currently the secondary NAAQS for NO_X and SO_X are being reviewed, specifically addressing the welfare effects of acidification and nitrogen nutrient enrichment.[X] As part of this review, ecosystem maps (Figures 3.3-3 through 3.3-6) [144] for the continental U.S. have been created that depict areas that are potentially sensitive to aquatic and terrestrial acidification, and aquatic and terrestrial nutrient enrichment. Taken together, these sensitive ecological areas are of greatest concern with regard to the deposition of nitrogen and sulfur compounds resulting from ship emissions. NO_X and SO_X emissions from ships today and in 2020 will significantly contribute to higher annual total nitrogen and sulfur deposition in all of these potentially sensitive ecosystems. See Section 3.3.1.7 for a discussion and accompanying maps which documents both the level and geographic impact of ship emissions in 2020 on nitrogen and sulfur deposition in the U.S.

Terrestrial Acidification-U.S. Geography

Deposition of total nitrogen (including both oxidized and reduced forms) and sulfur species contributing to acidification were routinely measured in the U.S. between 2004 and 2006 and those results are shown in Figures 3.3-1 and 3.3-2. Figure 3.3-3 depicts areas across the U.S. which are potentially sensitive to terrestrial acidification including forest ecosystems in the Adirondack Mountains located in the State of New York, the Green Mountains in the State of Vermont, the White Mountains in the State of New Hampshire, the Allegheny Plateau in the State of Pennsylvania, in the southeastern part of the U.S., and high-elevation ecosystems in the southern Appalachians. In addition, areas of the Upper Midwest and parts of the State of Florida are also at significant risk with regard to terrestrial acidification.

[X] The first draft risk and exposure assessment and other documents associated with this review are available at: http://www.epa.gov/ttn/naaqs/standards/no2so2sec/cr_rea.html

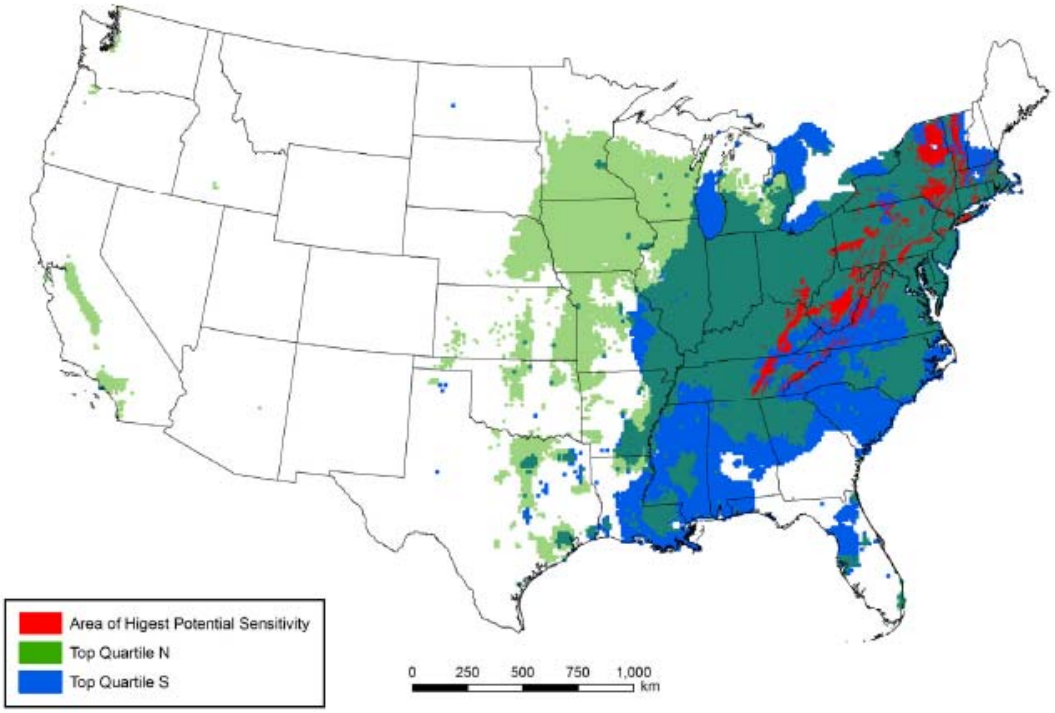


Figure 3.3-3 Areas Potentially Sensitive to Terrestrial Acidification

Aquatic Acidification-U.S. Geography

A number of national and regional assessments have been conducted to estimate the distribution and extent of surface water acidity in the U.S.[145,146,147,148,149,150,151,152,153] As a result, several regions of the U.S. have been identified as containing a large number of lakes and streams which are seriously impacted by acidification.

Figure 3.3-4 illustrates those areas of the U.S. where aquatic ecosystems are at risk from acidification. These sensitive ecological regions include portions of the Northeast U.S - especially all the New England States, the Adirondacks, and the Catskill Mountains in the State of New York; the Southeast U.S.-including the Appalachian Mountains and the northern section of the State of Florida; all upper Midwest States and parts of the western U.S.[154] – especially the Los Angeles Basin and surrounding area and the Sierra Nevada Mountains in the State of California. Two western mountain ranges with the greatest number of acid sensitive lakes[155] are the Cascade Mountains, stretching from northern California, through the entire States of Oregon and Washington, and the Sierra Nevada's, found within the State of California. The hydrologic cycles in these two mountain ranges are dominated by the annual accumulation and melting of a dilute, mildly acidic snow pack. Finally, also in the western U.S., many Rocky Mountain lakes in the State of Colorado are also sensitive to acidifying deposition effects.[156] However, it does not appear that chronic acidification has occurred to any significant degree in these lakes, although episodic acidification has been reported for some.[157]

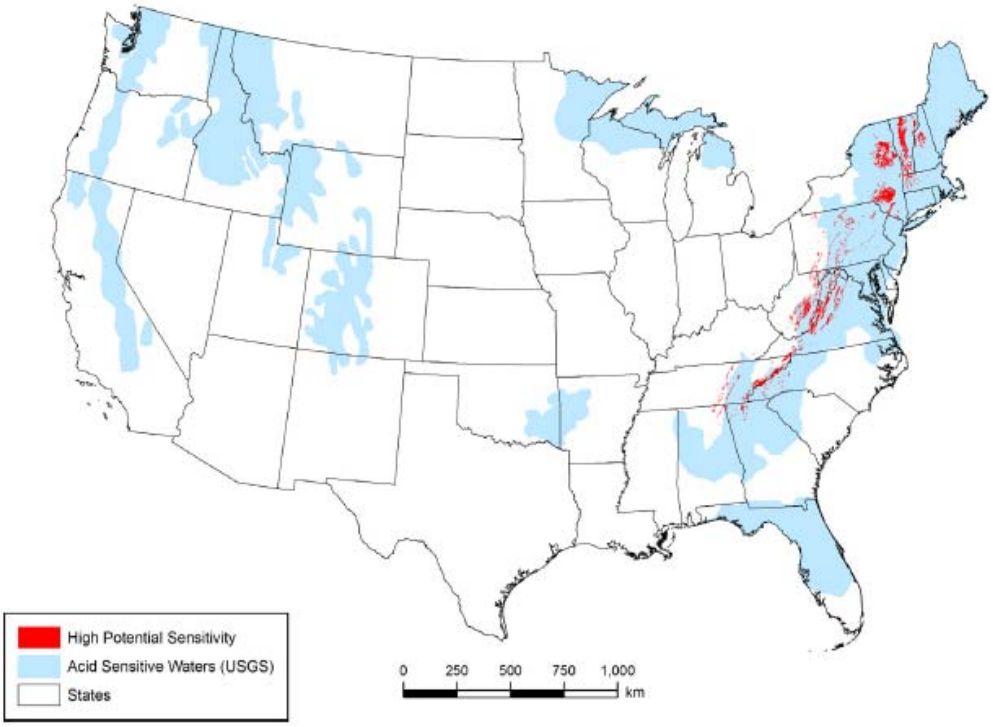


Figure 3.3-4 Areas Potentially Sensitive to Aquatic Acidification

Terrestrial Nutrient Enrichment-U.S. Geography

Nitrogen deposition affects terrestrial ecosystems throughout large areas of the U.S.[158] Atmospheric nitrogen deposition is the main source of new nitrogen in many terrestrial ecosystems throughout the U.S and impacts large numbers of forests, wetlands, freshwater bogs and salt marshes.[159] Figure 3.3-5 depicts those ecosystems potentially sensitive to terrestrial nutrient enrichment resulting from nitrogen deposition - including nitrogen deposition from ships.

Severe symptoms of nutrient enrichment or nitrogen saturation, have been observed in forest ecosystems of the State of West Virginia's northern hardwood watersheds;[160] in high-elevation spruce-fir ecosystems in the Appalachian Mountains;[161] in spruce-fir ecosystems throughout the northeastern U.S.;[162,163] and in lower-elevation eastern U.S. forests.[164,165,166,167] In addition, mixed conifer forests in the Los Angeles Air Basin within the State of California are also heavily impacted and exhibit the highest stream water nitrate concentrations documented within wild lands in North America.[168,169] In general, it is believed that deciduous forest stands in the eastern U.S. have not progressed toward nitrogen saturation as rapidly or as far as coniferous stands in the eastern U.S.[170]

In addition to these forest ecosystems, nitrogen deposition adversely impacts U.S. grasslands or prairies which are located throughout the U.S.[171] The vast majority of these grasslands are found in the Central Plains regions of the U.S. between the Mississippi River and the foothills of the Rocky Mountains. However, some native grasslands are scattered throughout the Midwestern and Southeastern U.S.[172] Also considered sensitive to nitrogen

nutrient enrichment effects, and receiving high levels of atmospheric deposition, are some arid and semi-arid ecosystems and desert ecosystems in the southwestern U.S.[173] However, water is generally more limiting than nitrogen in these areas. The alpine ecosystems in the State of Colorado, chaparral watersheds of the Sierra Nevada Mountains in the State of California, lichen and vascular plant communities in the San Bernardino Mountains in California and the entire U.S. Pacific Northwest, and the Southern California coastal sage scrub community are among the most sensitive terrestrial ecosystems to nitrogen deposition in the U.S. [174,175]

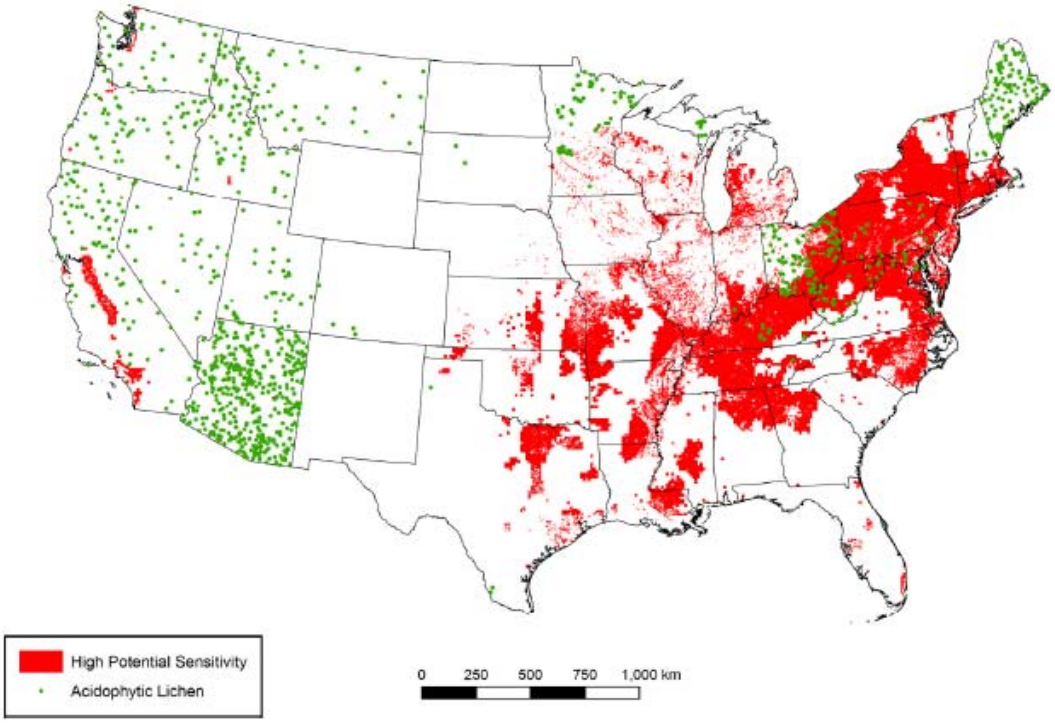


Figure 3.3-5 Areas Potentially Sensitive to Terrestrial Nutrient Enrichment

Aquatic Nutrient Enrichment –U.S. Geography

Aquatic nutrient enrichment impacts a wide range of waters within the U.S. from wetlands, to streams, rivers, lakes, estuaries and coastal waters. All are vital ecosystems to the U.S. and all are impacted by ship emissions that contribute to the annual total nitrogen deposition in the U.S.

Wetlands are found throughout the U.S. and support over 4200 native plant species, of which 121 have been designated by the U.S. government as threatened or endangered.[176] Freshwater wetlands are particularly sensitive to nutrient enrichment resulting from nitrogen deposition since they contain a disproportionately high number of rare plant species that have evolved under nitrogen-limited conditions.[177] Freshwater wetlands receive nitrogen mainly from precipitation, land runoff or ground water. Intertidal wetlands develop on sheltered coasts or in estuaries where they are periodically inundated by marine water that often carries high nitrogen loads, in addition to receiving water and nutrient inputs from precipitation and ground/surface water. Wetlands can be divided into three general categories based on

hydrology: (1) Peatlands and bogs, (2) fens, freshwater marshes, freshwater swamps and (3) intertidal wetlands.

Fens and bogs are the most vulnerable type of wetland ecosystems with regard to nutrient enrichment effects of nitrogen deposition.[178] In the U.S. they are mostly found in the glaciated northeast and Great Lakes regions and in the State of Alaska, but also in the southeast U.S. along the Atlantic Coastal Plain stretching from the States of Virginia through North Carolina to northern Florida.[179] Like bogs, fens are mostly a northern hemisphere phenomenon -- occurring in the northeastern United States, the Great Lakes region, western Rocky Mountains, and much of Canada -- and are generally associated with low temperatures and short growing seasons, where ample precipitation and high humidity cause excessive moisture to accumulate.[180]

The third type of wetlands sensitive to nitrogen deposition are marshes, characterized by emergent soft-stemmed vegetation adapted to saturated soil conditions. There are many different kinds of marshes in the U.S., ranging from the prairie potholes in the interior of the U.S. to the Everglades found in the extreme southern portion of the State of Florida. U.S. fresh water marshes are important for recharging groundwater supplies, and moderating stream flow by providing water to streams and as habitats for many wildlife species.[181]

Nitrogen deposition is the main source of nitrogen for many surface waters in the U.S. including headwater streams, lower order streams, and high elevation lakes.[182,183] Elevated surface water nitrate concentrations due to nitrogen deposition occur in both the eastern and western U.S. although high concentrations of nitrate in surface waters in the western U.S. are not as widespread as in the eastern U.S.

High concentrations of lake or stream water nitrate, indicative of ecosystem nitrogen-saturation, have been found at a variety of locations throughout the U.S. including the San Bernardino and San Gabriel Mountains within the Los Angeles Air Basin in the State of California [184], the Front Range Mountains in the State of Colorado,[185,186] the Allegheny Mountains in the State of West Virginia,[187] the Catskill and Adirondack Mountains in the State of New York,[188, 189,190] and the Great Smoky Mountains in the State of Tennessee.

Nitrogen nutrient enrichment is a major environmental problem facing all U.S. coastal regions, but especially the Eastern, mid-Atlantic, and Gulf Coast regions, as excess nitrogen leads to eutrophication. There is broad scientific consensus that nitrogen-driven eutrophication of shallow estuaries in the U.S. has increased over the past several decades and that environmental degradation of coastal ecosystems is now a widespread occurrence.[191] A recent national assessment of eutrophic conditions in U.S. estuaries found that 65% of the assessed systems had moderate to high overall eutrophic conditions.[192] Estuaries and coastal waters tend to be nitrogen-limited and are therefore inherently sensitive to increased atmospheric nitrogen deposition.[193] Of 138 estuaries examined in the National Assessment, 44 were identified as showing symptoms of nutrient enrichment. Of the 23 estuaries examined in the Northeast U.S. 61% were classified as moderately to severely degraded. Other regions of the U.S. had mixtures of low, moderate, and high degree of eutrophication.[194] The contribution from atmospheric nitrogen deposition can be greater than

30% of total nitrogen loads in some of the most highly eutrophic estuaries in the US, including the Chesapeake Bay.

EPA's draft risk and exposure assessment (REA) for the NOxSOx secondary NAAQS developed an overview map of the U.S. that identifies areas of national aquatic nutrient enrichment sensitivity. They utilized the eutrophic estuaries from NOAA's Coastal Assessment Framework and areas that exceed the nutrient criteria for lakes/reservoirs (U.S. EPA, 2002). Both these were combined and compared to total nitrogen deposition. The resulting map revealed areas of highest potential sensitivity to nitrogen deposition as shown in Figure 3.3-6. These areas are identified in blue as nutrient sensitive estuaries contained in NOAA's Coastal Assessment Framework (CAF), and red in areas where deposition exceeds the nutrient criteria. Yellow areas indicate those areas that are below the nutrient criteria but are within 5 kg N/ha/yr of exceeding it.

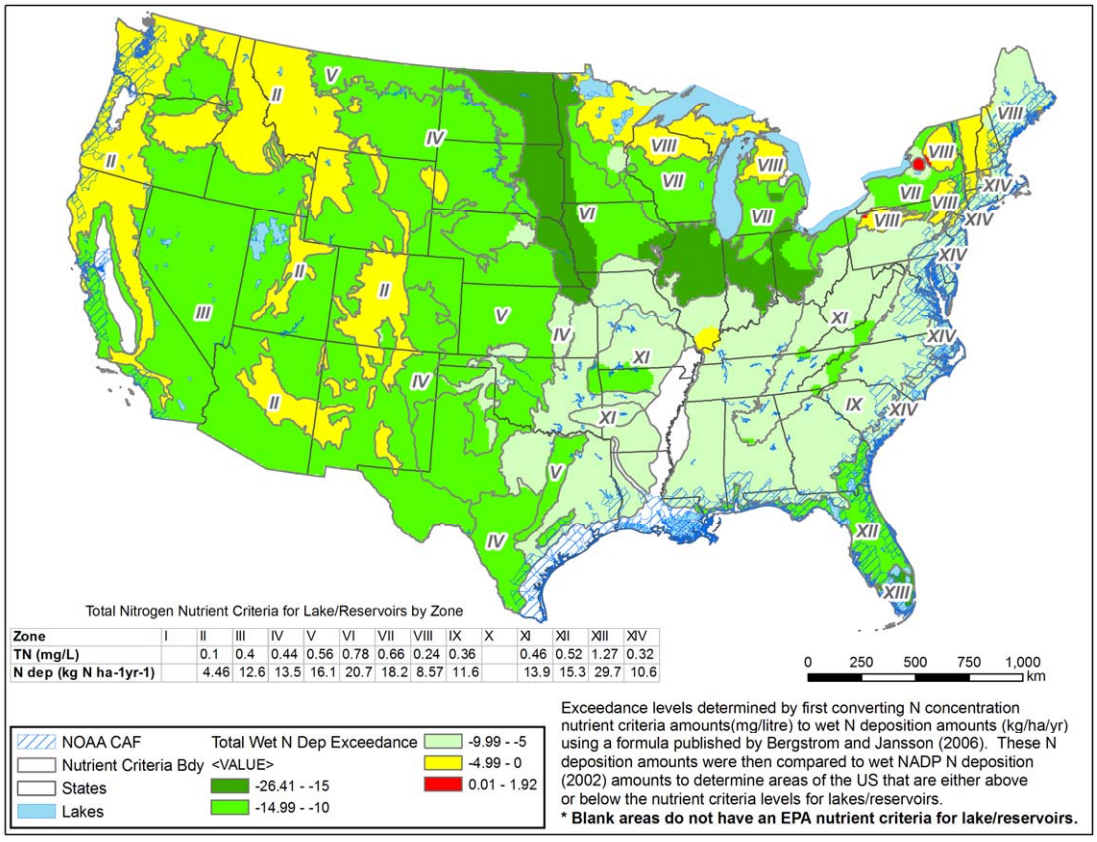


Figure 3.3-6 Areas Potentially Sensitive to Aquatic Nutrient Enrichment

The most extreme effects of nitrogen deposition on U.S. aquatic ecosystems result in severe nitrogen-loading to these ecosystems that contribute to hypoxic zones devoid of life**.** Three hypoxia zones of special concern in the U.S. are (1) the zone located in the Gulf of Mexico straddling the States of Louisiana and Texas, (2) The Chesapeake Bay located between the States of Maryland and Virginia, and (3) Long Island Sound located between the States of New York and Connecticut. The largest hypoxia zone in the U.S. is in the northern Gulf of Mexico along the continental shelf. During midsummer, this zone has regularly been larger than 16,000km^2.[195] Figure 3.3-7 depicts the location of these three hypoxic zones.

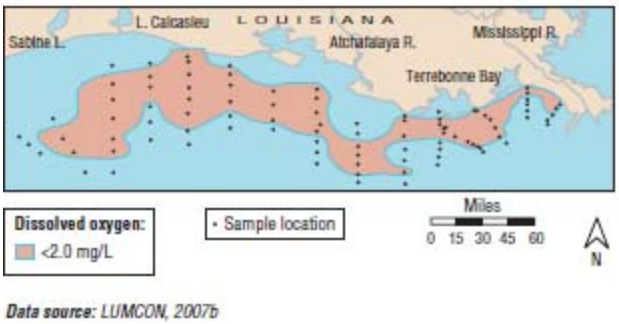


Figure 3.3-7 a. Hypoxia Zone in 2007 for the Gulf of Mexico

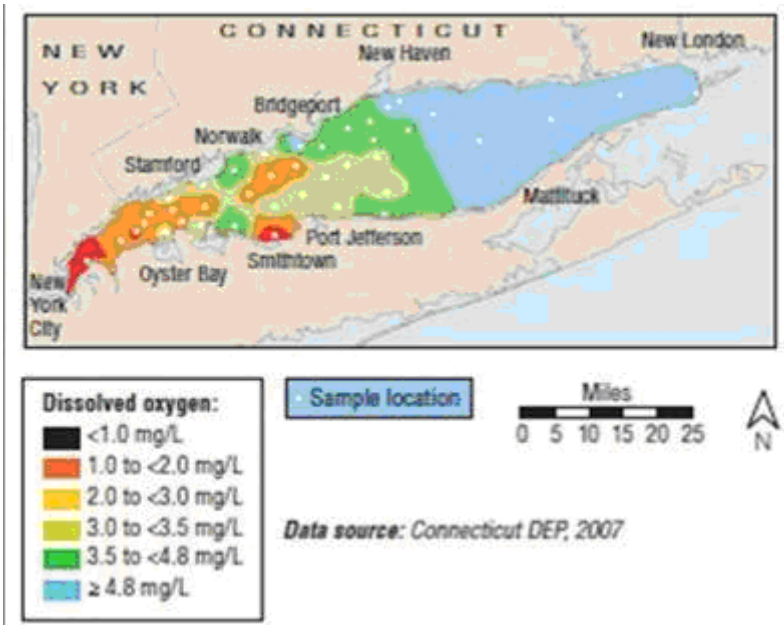
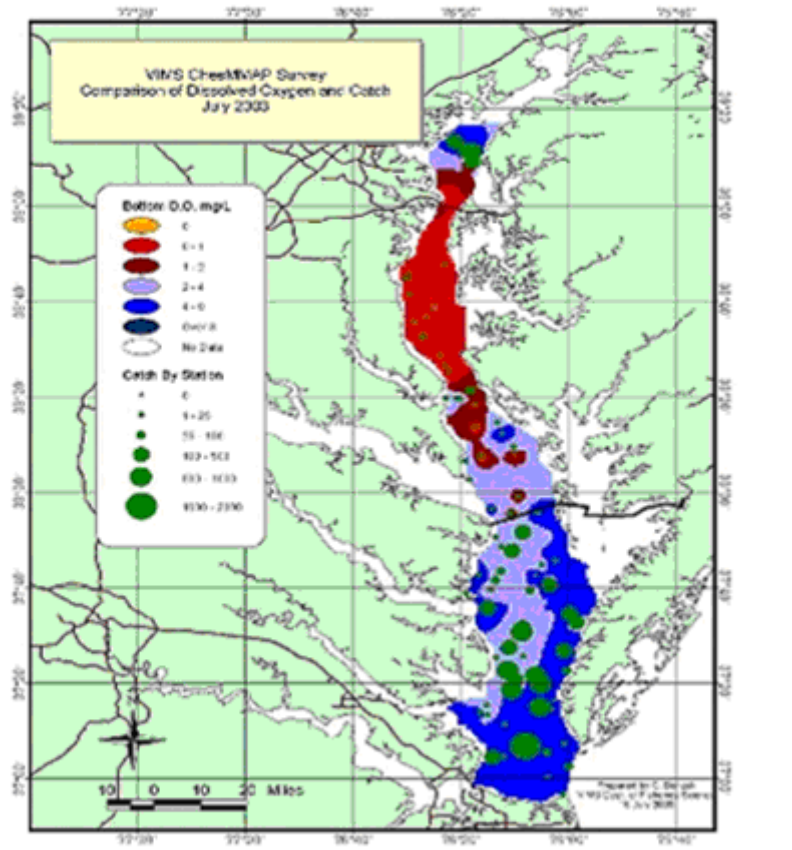


Figure 3.3-7 b. Hypoxia Zone in 2007 for Long Island Sound



Varying dissolved oxygen levels and overall fish catch in the Chesapeake Bay through July, 2003. Source: Virginia Institute of Marine Science

Figure 3.3-7 c Hypoxia Zone for Chesapeake Bay in 2003

3.3.1.3 Science of Nitrogen and Sulfur Deposition

Nitrogen and sulfur interactions in the environment are highly complex. Both are essential, and sometimes limiting, nutrients needed for growth and productivity. Excess of nitrogen or sulfur can lead to acidification, nutrient enrichment, and eutrophication.

Ships release emissions over a wide area, and depending on prevailing winds and other meteorological conditions, these emissions may be transported hundreds and even thousands of kilometers across North America. Section 3.2 discusses the results of U.S. air quality modeling which documents this phenomenon. Overall, these engines emit a large amount of NO_X, SO_X and direct PM which impact not only ambient air concentrations but also contribute to deposition of nitrogen and sulfur in many sensitive ecological areas throughout the U.S.

The sulfur in marine fuel is primarily emitted as sulfur dioxide (SO_2), with a small fraction (about two percent) being converted to sulfur trioxide (SO_3).[196] SO_3 almost immediately forms sulfate and is also emitted as primary PM by the engine and consists of carbonaceous material, sulfuric acid, and ash (trace metals). The vast majority of the primary PM is less than or equal to 2.5 μm in diameter, and accounts for the majority of the number of

particles in exhaust, but only a small fraction of the mass of diesel PM. These particles also react in the atmosphere to form secondary PM, which exist there as a carbon core with a coating of organic carbon compounds, nitrate particles, or as sulfuric acid and ash, sulfuric acid aerosols, or sulfate particles associated with organic carbon.

At the same time, ships emit large amounts of nitric oxide (NO) and nitrogen dioxide (NO_2) emissions which are carried into the atmosphere where they may be chemically altered and transformed into new compounds. For example, NO_2 can also be further oxidized to nitric acid (HNO_3) and can contribute in that form to the acidity of clouds, fog, and rain water and can also form ambient particulate nitrate (pNO_3) which may be deposited either directly onto terrestrial and aquatic ecosystems ("direct deposition") or deposited onto land surfaces where it subsequently runs off and is transferred into downstream waters ("indirect deposition").

Deposition of nitrogen and Sulfur resulting from ship operations can occur either in a wet or dry form. Wet deposition includes rain, snow, sleet, hail, clouds, or fog. Dry deposition includes gases, dust, and minute particulate matters. Wet and dry atmospheric deposition of $PM_{2.5}$ delivers a complex mixture of metals (such as mercury, zinc, lead, nickel, arsenic, aluminum, and cadmium), organic compounds (such as polycyclic organic matter, dioxins, and furans) and inorganic compounds (such as nitrate, sulfate) to terrestrial and aquatic ecosystems.

The chemical form of deposition is determined by ambient conditions (e.g., temperature, humidity, oxidant levels) and the pollutant source. Chemical and physical transformations of ambient particles occur in the atmosphere and in the media (terrestrial or aquatic) on which they deposit. These transformations influence the fate, bioavailability and potential toxicity of these compounds. The atmospheric deposition of metals and toxic compounds is implicated in severe ecosystem effects.[197]

Ships also emit primary PM. In addition, secondary PM is formed from NO_X and SO_X gaseous emissions and associated chemical reactions in the atmosphere. The major constituents of secondary PM are sulfate, nitrate, ammonium, and hydrogen ions. Secondary aerosol formation depends on numerous factors including the concentrations of precursors; the concentrations of other gaseous reactive species such as ozone, hydroxyl radical, peroxy radicals, and hydrogen peroxide; atmospheric conditions, including solar radiation and relative humidity; and the interactions of precursors and preexisting particles within cloud or fog droplets or on or in the liquid film on solid particles.[198]

The lifetimes of particles vary with particle size. Accumulation-mode particles such as the sulfates and nitrates are kept in suspension by normal air motions and have a lower deposition velocity than coarse-mode particles; they can be transported thousands of kilometers and remain in the atmosphere for a number of days. They are removed from the atmosphere primarily by cloud processes. Dry deposition rates are expressed in terms of deposition velocity that varies with the particle size, reaching a minimum between 0.1 and 1.0 micrometer (μm) aerodynamic diameter.[199]

Particulate matter is a factor in acid deposition. Particles serve as cloud condensation nuclei and contribute directly to the acidification of rain. In addition, the gas-phase species that lead to the dry deposition of acidity are also precursors of particles. Therefore, reductions in NO_X and SO_2 emissions will decrease both acid deposition and PM concentrations, but not necessarily in a linear fashion. Sulfuric acid, ammonium nitrate, and organic particles also are deposited on surfaces by dry deposition and can contribute to environmental effects.[200]

3.3.1.4 Computing Atmospheric Nitrogen and Sulfur Deposition to Specific Locations

Inputs of new nitrogen, i.e., non-recycled mostly anthropogenic in origin, are often key factors controlling primary productivity in nitrogen-sensitive estuarine and coastal waters.[201] Increasing trends in urbanization, agricultural intensity, and industrial expansion have led to increases in nitrogen deposited from the atmosphere on the order of a factor of 10 in the previous 100 years.[202] Direct fluxes of atmospheric nitrogen to ocean and gulf waters along the northeast and southeast U.S. are now roughly equal to or exceed the load of new nitrogen from riverine inputs at 11, 5.6, and 5.6 kg N/ha for the northeast Atlantic coast of the U.S., the southeast Atlantic coast of the U.S., and the U.S. eastern Gulf of Mexico, respectively.[203] Atmospheric nitrogen is dominated by a number of sources, most importantly transportation sources, including ships.

Nitrogen deposition takes different forms physically. Physically, deposition can be direct, with the loads resulting from air pollutants depositing directly to the surface of a body of water, usually a large body of water like an estuary or lake. In addition, there is an indirect deposition component derived from deposition of nitrogen or sulfur to the rest of the watershed, both land and water, of which some fraction is transported through runoff, rivers, streams, and groundwater to the water body of concern.

Direct and indirect deposition of nitrogen and sulfur to watersheds depend on air pollutant concentrations in the airshed above the watershed. The shape and extent of the airshed is quite different from that of the watershed. In a watershed, everything that falls in its area, by definition, flows into a single body of water. An airshed, by contrast, is a theoretical concept that defines the source area containing the emissions contributing a given level, often 75%, to the deposition in a particular watershed or to a given water body. Hence, airsheds are modeled domains containing the sources estimated to contribute a given level of deposition from each pollutant of concern. The principal NO_X airsheds and corresponding watersheds for several regions in the eastern U.S. are shown in Figure 3.3-8.[204] These airsheds extend well into U.S. coastal waters where ships operate.

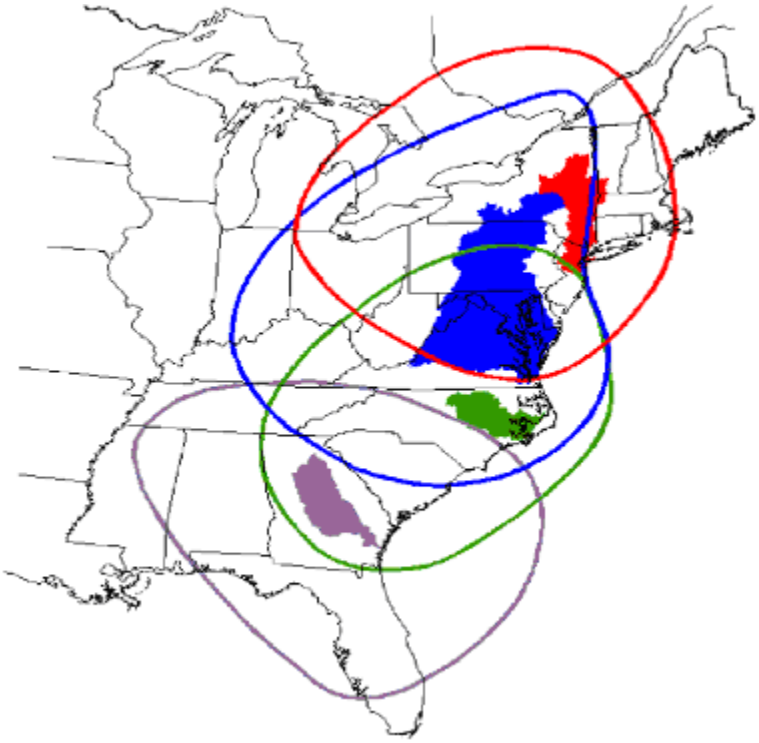


Figure 3.3-8 Principal Airsheds and Watersheds for Oxides of Nitrogen for Estuaries.Hudson/Raritan Bay; Chesapeake Bay; Pamlico Sound; and Altamaha Sound (listed from north to south).

Nitrogen inputs have been studied in several U.S. Gulf Coast estuaries, as well, owing to concerns about eutrophication there. Nitrogen from atmospheric deposition in these locations is estimated to be 10 to 40% of the total input of nitrogen to many of these estuaries, and could be higher for some. Estimates of total nitrogen loadings to estuaries or to other large-scale elements in the landscape are then computed using measurements of wet and dry deposition, where these are available, and interpolated with or without a set of air quality model predictions such as the Extended Regional Acid Deposition Model (Ext-RADM).[205,206,207,208,209]

Table 3.3-2 lists several water bodies for which atmospheric nitrogen inputs have been computed and the ratio to total nitrogen loads is given. The contribution from the atmosphere ranges from a low of 2–8% for the Guadalupe Estuary in the southern part of the State of Texas to highs of ~38% in the New York State Bight and the Albemarle-Pamlico Sound in the State of North Carolina.

Table 3.3-2 Atmospheric Nitrogen Loads Relative to Total Nitrogen Loads in Selected U.S. Great

Waters*

Waterbody	Total N Load (million kg/yr)	Atmospheric N Load (million kg/yr)	Percent Load from the Atmosphere
Albemarle-Pamlico Sounds	23	9	38
Chesapeake Bay	170	36	21
Delaware Bay	54	8	15
Long Island Sound	60	12	20
Narragansett Bay	5	0.6	12
New York Bight	164	62	38
Based on ADN N loads from the watershed only (excluding direct N deposition to the bay surface):			
Waquoit Bay, MA	0.022	0.0065	29
Based on ADN directly to the waterbody (excluding ADN loads from the watershed):			
Delaware Inland Bays	1.3	0.28	21
Flanders Bay, NY	0.36	0.027	7
Guadalupe Estuary, TX	4.2–15.9	0.31	2–8
Massachusetts Bays	22–30	1.6–6	5–27
Narragansett Bay	9	0.4	4
Newport River Coastal Waters, NC	0.27–0.85	0.095–0.68	>35
Potomac River, MD	35.5	1.9	5
Sarasota Bay, FL	0.6	0.16	26
Tampa Bay, FL	3.8	1.1	28

ADN = atmospheric deposition of N

Source: *Table from Deposition of Air Pollutants to the Great Waters-3rd Report to Congress (EPA, 2000)

3.3.1.5 Summary of Ecological Effects Associated with Nitrogen and Sulfur and PM Deposition

Deposition of reduced and oxidized nitrogen and sulfur species cause acidification, altering biogeochemistry and affecting animal and plant life in terrestrial and aquatic ecosystems across the U.S. Major effects include a decline in sensitive tree species, such as red spruce and sugar maple; and a loss of biodiversity of fishes, zooplankton, and macro invertebrates. The sensitivity of terrestrial and aquatic ecosystems to acidification from nitrogen and sulfur deposition is predominantly governed by the earth's geology.

Biological effects of acidification in terrestrial ecosystems are generally linked to aluminum toxicity and decreased ability of plant roots to take up base cations. Decreases in acid neutralizing capacity and increases in inorganic aluminum concentration contribute to declines in zooplankton, macro invertebrates, and fish species richness in aquatic ecosystems. Across the U.S., ecosystems continue to be acidified by current emissions from both stationary sources, area sources, and mobile sources. For example, in the Adirondack Mountains of New York State, the current rates of nitrogen and sulfur deposition exceed the amount that would allow recovery of the most acid sensitive lakes to a sustainable acid neutralizing capacity (ANC) level.[210]

In addition to the role nitrogen deposition plays in acidification, it also causes ecosystem nutrient enrichment and eutrophication that alters biogeochemical cycles and harms animal and plant life such as native lichens and alters biodiversity of terrestrial ecosystems, such as forests and grasslands. Nitrogen deposition contributes to eutrophication of estuaries and coastal waters which result in toxic algal blooms and fish kills. For example, the Chesapeake Bay Estuary is highly eutrophic and 21 -30% of total nitrogen load comes from deposition. Freshwater ecosystems may also be impacted by nitrogen deposition, for example, high elevation freshwater lakes in the western U.S. experience adverse ecosystem changes at nitrogen deposition rates as low as 2 kg N/ha/yr.[211]

The addition of nitrogen to most ecosystems causes changes in primary productivity and growth of plants and algae, which can alter competitive interactions among species. Some species grow more than others, leading to shifts in population dynamics, species composition, and community structure. The most extreme effects of nitrogen deposition include a shift of ecosystem types in terrestrial ecosystems, and hypoxic zones that are devoid of life in aquatic ecosystems**.**[212]

There are a number of important quantified relationships between nitrogen deposition levels and ecological effects. Certain lichen species are the most sensitive terrestrial taxa to nitrogen with species losses occurring at just 3 kg N/ha/yr in the U.S. Pacific Northwest and in the southern portion of the State of California. The onset of declining biodiversity was found to occur at levels of 5 kg N/ha/yr and above within grasslands in both the State of Minnesota and in Europe. Altered species composition of Alpine ecosystems and forest encroachment into temperate grasslands was found at 10 kg N/ha/yr and above in both the U.S.[213]

A United States Forest Service study conducted in areas within the Tongass Forest in Southeast Alaska found evidence of sulfur emissions impacting lichen communities. The authors concluded that the main source of sulfur and nitrogen found in lichens from Mt. Roberts is likely the burning of fossil fuels by cruise ships and other vehicles and equipment in downtown Juneau.[214]

Lichen are an important food source for caribou. This is causing concern about the potential role damage to lichens may be having on the Southern Alaska Peninsula Caribou Herd,[215] which is an important food source to local subsistence based cultures. This herd has been decreasing in size, exhibiting both poor calf survival and low pregnancy rates, which are signs of dietary stress. Currently there is a complete caribou hunting ban, including a ban on subsistence hunting. If regulation of marine fuels could potentially enhance lichen biomass in the area, it would contribute in turn to maintenance of an important subsistence resource for local human populations.

The biogeochemical cycle of mercury, a well-known neurotoxin, is closely tied to the sulfur cycle. Mercury is taken up by living organisms in the methylated form, which is easily bioaccumulated in the food web. Sulfate-reducing bacteria in wetland and lake sediments play a key role in mercury methylation. Changes in sulfate deposition have resulted in changes in both the rate of mercury methylation and the corresponding mercury

concentrations in fish. In 2006, 3,080 fish advisories were issued in the U.S. due to the presence of methyl mercury in fish.[216]

Although sulfur deposition is important to mercury methylation, several other interrelated factors seem to also be related to mercury uptake, including low lake water pH, dissolved organic carbon, suspended particulate matter concentrations in the water column, temperature, and dissolved oxygen. In addition, the proportion of upland to wetland land area within a watershed, as well as wetland type and annual water yield, appear to be important.

Current international shipping emissions of $PM_{2.5}$ contain small amounts of metals—nickel, vanadium, cadmium, iron, lead, copper, zinc, aluminum.[217,218,219] Investigations of trace metals near roadways and industrial facilities indicate that a substantial burden of heavy metals can accumulate on vegetative surfaces. Copper, zinc, and nickel are shown to be directly toxic to vegetation under field conditions.[220] While metals typically exhibit low solubility, limiting their bioavailability and direct toxicity, chemical transformations of metal compounds occur in the environment, particularly in the presence of acidic or other oxidizing species. These chemical changes influence the mobility and toxicity of metals in the environment. Once taken up into plant tissue, a metal compound can undergo chemical changes, accumulate and be passed along to herbivores or can re-enter the soil and further cycle in the environment.

Although there has been no direct evidence of a physiological association between tree injury and heavy metal exposures, heavy metals have been implicated because of similarities between metal deposition patterns and forest decline.[221] This hypothesized correlation was further explored in high elevation forests in the northeastern U.S. These studies measured levels of a group of intracellular compounds found in plants that bind with metals and are produced by plants as a response to sublethal concentrations of heavy metals. These studies indicated a systematic and significant increase in concentrations of these compounds associated with the extent of tree injury. These data strongly imply that metal stress causes tree injury and contributes to forest decline in Northeast U.S.[222] Contamination of plant leaves by heavy metals can lead to elevated soil levels. Trace metals absorbed into the plant frequently bind to the leaf tissue, and then are lost when the leaf drops. As the fallen leaves decompose, the heavy metals are transferred into the soil.[223, 224]

Ships also emit air toxics, including polycyclic aromatic hydrocarbons (PAHs) -- a class of polycyclic organic matter (POM) that contain compounds which are known or suspected carcinogens. Since the majority of PAHs are adsorbed onto particles less than 1.0 μm in diameter, long range transport is possible. Particles of this size can remain airborne for days or even months and travel distances up to 10,000km before being deposited on terrestrial or aquatic surfaces.[225] Atmospheric deposition of particles is believed to be the major source of PAHs to the sediments of Lake Michigan in the Great Lakes, Chesapeake Bay, which is surrounded by the States of Maryland and Virginia, Tampa Bay in the central part of the State of Florida and in other coastal areas of the U.S.[226,227,228,229,230] PAHs tend to accumulate in sediments and reach high enough concentrations in some coastal environments to pose an environmental health threat that includes cancer in fish populations, toxicity to organisms living in the sediment and risks to those (e.g., migratory birds) that consume these

organisms.[231, 232] PAHs tend to accumulate in sediments and bioaccumulate in freshwater, flora and fauna.

3.3.1.6 Ecological Effects Nutrient Enrichment

In general, ecosystems that are most responsive to nutrient enrichment from atmospheric nitrogen deposition are those that receive high levels of nitrogen loading, are nitrogen-limited, or contain species that have evolved in nutrient-poor environments. Species that are adapted to low nitrogen supply will often be more readily outcompeted by species that have higher nitrogen demands when the availability of nitrogen is increased.[233,234, 235,236] As a consequence, some native species can be eliminated by nitrogen deposition.[237,238,239, 240] Note the terms "low" and "high" are relative to the amount of bioavailable nitrogen in the ecosystem and the level of deposition.

Eutrophication effects resulting from excess nitrogen are more widespread than acidification effects in western North America. Figure 3.3-9 highlights areas in the Western U.S. where nitrogen effects have been extensively reported. The discussion of ecological effects of nutrient enrichment which follows is organized around three types of ecosystem categories which experience impacts from nutrient enrichment: terrestrial, transitional, and aquatic.

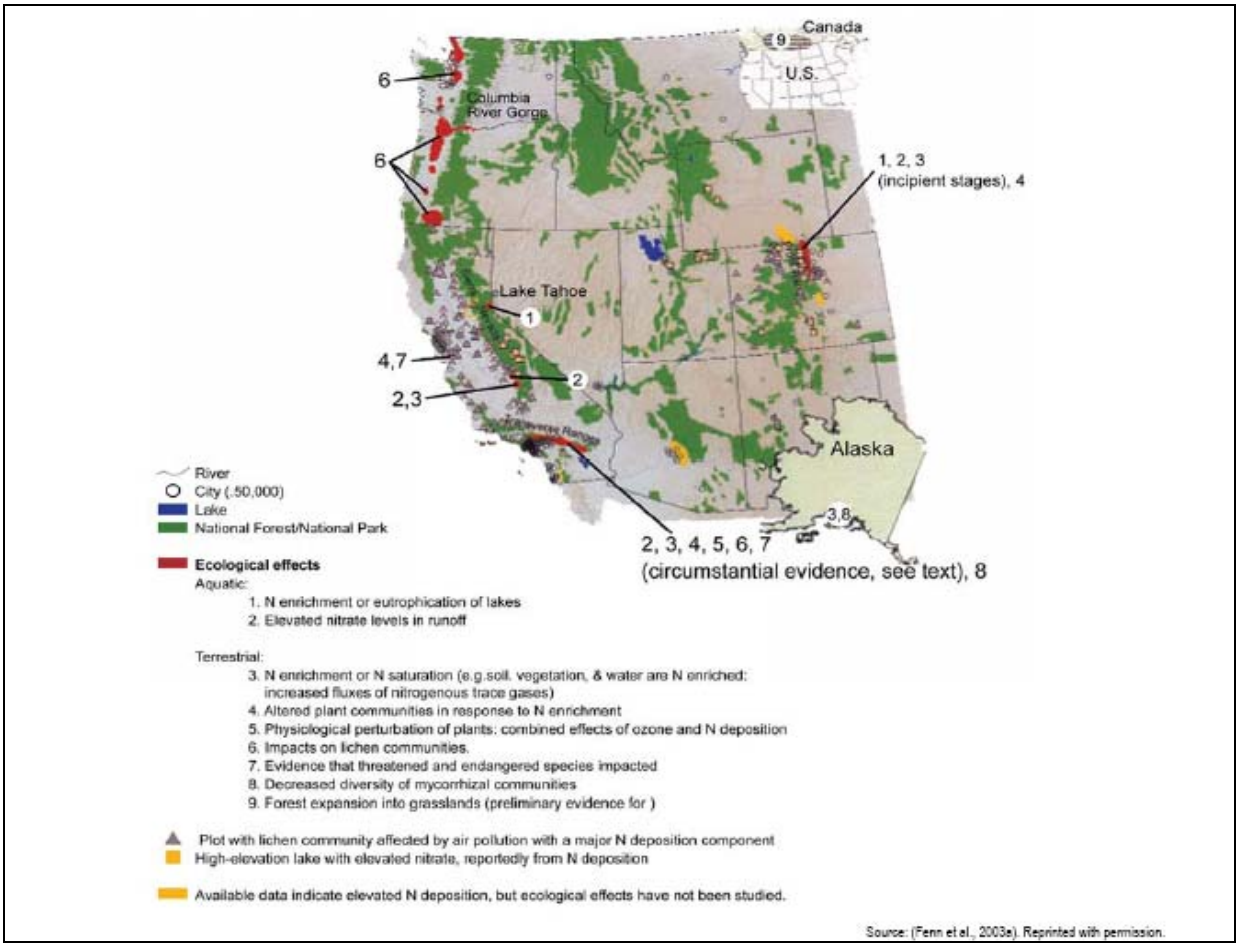


Figure 3.3-9. Map of the Western U.S. Showing the Primary Geographic Areas where Nitrogen Deposition Effects have been Reported

Terrestrial

Ecological effects of nitrogen deposition occur in a variety of taxa and ecosystem types including: forests, grasslands, arid and semi-arid areas, deserts, lichens, alpine, and mycorrhizae. Atmospheric inputs of nitrogen can alleviate deficiencies and increase growth of some plants at the expense of others. Nitrogen deposition alters the competitive relationships among terrestrial plant species and therefore alters species composition and diversity.[241,242,243] Wholesale shifts in species composition are easier to detect in short-lived terrestrial ecosystems such as annual grasslands, in the forest understory, or mycorrhizal associations, than for long-lived forest trees where changes are evident on a decade or longer time scale. Note species shifts and ecosystem changes can occur even if the ecosystem does not exhibit signs of nitrogen saturation.

There are a number of important quantified relationships between nitrogen deposition levels and ecological effects.[244] Certain lichen species are the most sensitive terrestrial taxa to nitrogen in the U.S. with clear adverse effects occurring at just 3 kg N/ha/yr. Figure 3-5 shows the geographic distribution of lichens in the U.S. Among the most sensitive U.S. ecosystems are Alpine ecosystems where alteration of plant covers of an individual species (*Carex rupestris*) was estimated to occur at deposition levels near 4 kg N/ha/yr and modeling indicates that deposition levels near 10 kg/N/ha/yr alter plant community assemblages.[245] Within grasslands, the onset of declining biodiversity was found to occur at levels of 5 kg N/ha/yr. Forest encroachment into temperate grasslands was found at 10 kg N/ha/yr and above in the U.S. Table 3.3-3 provides a brief list of nitrogen deposition levels and associated ecological effects.

Table 3.3-3 Examples of Quantified Relationship Between Nitrogen Deposition Levels and Ecological Effects[a]

Kg N/ha/yr	Ecological effect
~1.5	Altered diatom communities in high elevation freshwater lakes and elevated nitrogen in tree leaf tissue high elevation forests in the U.S.
3.1	Decline of some lichen species in the Western U.S. (critical load)
4	Altered growth and coverage of alpine plant species in U.S.
5	Onset of decline of species richness in grasslands of the U.S. and U.K.
5.6 - 10	Onset of nitrate leaching in Eastern forests of the U.S.
5-10	Multiple effects in tundra, bogs and freshwater lakes in Europe (critical loads)
5-15	Multiple effects in arctic, alpine, subalpine and scrub habitats in Europe (critical loads)

Note: [a] EPA, Integrated Science Assessment for Oxides of Nitrogen and Sulfur-Ecological criteria

Most terrestrial ecosystems are nitrogen-limited, therefore they are sensitive to perturbation caused by nitrogen additions.[246] The factors that govern the vulnerability of terrestrial ecosystems to nutrient enrichment from nitrogen deposition include the degree of nitrogen limitation, rates and form of nitrogen deposition, elevation, species composition, length of growing season, and soil nitrogen retention capacity.

Regions and ecosystems in the western U.S. where nitrogen nutrient enrichment effects have been documented in terrestrial ecosystems are shown on Figure 3.3-9.[247] The alpine ecosystems of the Colorado Front Range, chaparral watersheds of the Sierra Nevada, lichen and vascular plant communities in the San Bernardino Mountains and the Pacific Northwest, and the southern California coastal sage scrub community are among the most sensitive terrestrial ecosystems in the western U.S.

In the eastern U.S., the degree of nitrogen saturation of the terrestrial ecosystem is often assessed in terms of the degree of nitrate leaching from watershed soils into ground water or surface water. Studies have estimated the number of surface waters at different stages of saturation across several regions in the eastern U.S.[248] Of the 85 northeastern watersheds examined, 40% were in nitrogen-saturation Stage 0[Y], 52% in Stage 1, and 8% in Stage 2. Of the northeastern sites for which adequate data were available for assessment, those in Stage 1 or 2 were most prevalent in the Adirondack and Catskill Mountains in the State of New York.

Transitional

About 107.7 million acres of wetlands are widely distributed in the conterminous U.S., 95 percent of which are freshwater wetlands and 5 percent are estuarine or marine wetlands[249] (Figure 3.3-10). At one end of the spectrum, bogs or peatland are very sensitive to nitrogen deposition because they receive nutrients exclusively from precipitation, and the species in them are adapted to low levels of nitrogen.[250, 251,252] Intertidal wetlands are at the other end of the spectrum; in these ecosystems marine/estuarine water sources generally exceed atmospheric inputs by one or two orders of magnitude.[253] Wetlands are widely distributed, including some areas that receive moderate to high levels of nitrogen deposition.

Nitrogen deposition alters species richness, species composition and biodiversity in U.S. wetland ecosystems.[254] The effect of nitrogen deposition on these ecosystems depends on the fraction of rainfall in its total water budget. Excess nitrogen deposition can cause shifts in wetland community composition by altering competitive relationships among species,

[Y] In Stage 0, nitrogen inputs are low and there are strong nitrogen limitations on growth. Stage 1 is characterized by high nitrogen rentention and fertilization effect of added nitrogen on tree growth. Stage 2 includes the induction of nitrification and some nitrate leaching, though growth may still be high. In Stage 3 tree growth declines, nitrification and nitrate loss continue to increase, but nitrogen mineralization rates begin to decline.

which potentially leads to effects such as decreasing biodiversity, increasing non-native species establishment and increasing the risk of extinction for sensitive and rare species.

U.S. wetlands contain a high number of rare plant species.[255,256, 257] High levels of atmospheric nitrogen deposition increase the risk of decline and extinction of these species that are adapted to low nitrogen conditions. In general these include the genus *Isoetes sp*., of which three species are federally endangered; insectivorous plants like the endangered green pitcher *Sarracenia oreophila*; and the genus *Sphagnum*, of which there are 15 species listed as endangered by eastern U.S. States. Roundleaf sundew (*Drosera rotundifolia*) is also susceptible to elevated atmospheric nitrogen deposition.[258] This plant is native to, and broadly distributed across, the U.S. and is federally listed as endangered in Illinois and Iowa, threatened in Tennessee, and vulnerable in New York.[259] In the U.S., *Sarracenia purpurea* can be used as a biological indicator of local nitrogen deposition in some locations.[260]

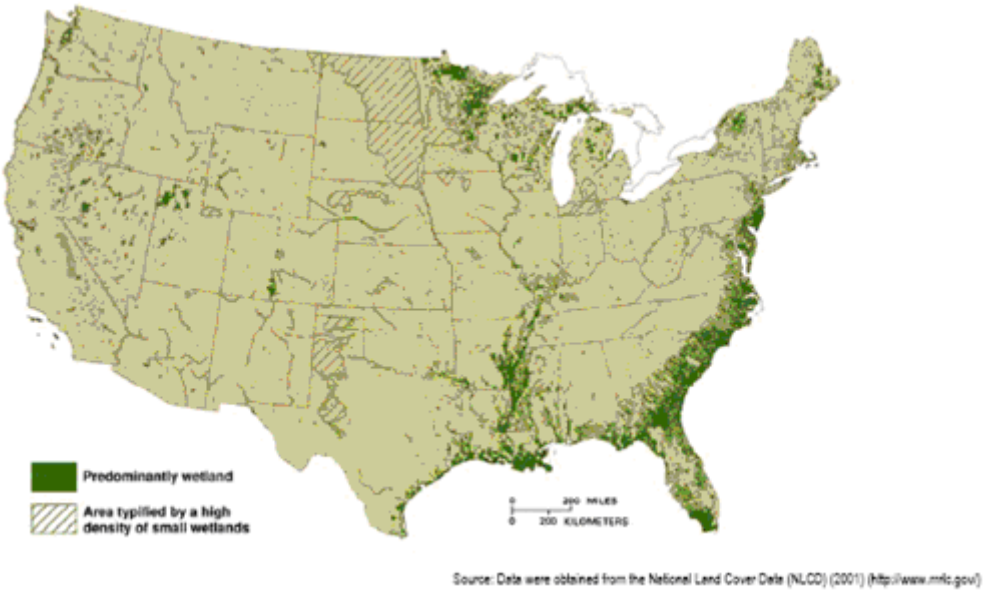


Figure 3.3-10 Location of Wetlands in Continental U.S.

Freshwater Aquatic

Nitrogen deposition alters species richness, species composition and biodiversity in freshwater aquatic ecosystems across the U.S.[261] Evidence from multiple lines of research and experimental approaches support this observation, including paleolimnological reconstructions, bioassays, mesocosm and laboratory experiments. Increased nitrogen deposition can cause a shift in community composition and reduce algal biodiversity. Elevated nitrogen deposition results in changes in algal species composition, especially in sensitive oligotrophic lakes. In the West, a hindcasting exercise determined that the change in Rocky Mountain National Park lake algae that occurred between 1850 and 1964 was associated with an increase in wet nitrogen deposition that was only about 1.5 kg N/ha. Similar changes inferred from lake sediment cores of the Beartooth Mountains of Wyoming also occurred at about 1.5 kg N/ha deposition.[262]

Some freshwater algae are particularly sensitive to added nutrient nitrogen and experience shifts in community composition and biodiversity with increased nitrogen deposition. For example, two species of diatom (a taxanomic group of algae), *Asterionella formosa* and *Fragilaria crotonensis*, now dominate the flora of at least several alpine and montane Rocky Mountain lakes. Sharp increases have occurred in Lake Tahoe.[263,264,265,266,267,268] The timing of this shift has varied, with changes beginning in the 1950s in the southern Rocky Mountains and in the 1970s or later in the central Rocky Mountains. These species are opportunistic algae that have been observed to respond rapidly to disturbance and slight nutrient enrichment in many parts of the world.

Estuarine Aquatic

Nitrogen deposition also alters species richness, species composition and biodiversity in estuarine ecosystems throughout the U.S.[269] Nitrogen is an essential nutrient for estuarine and marine fertility. However, excessive nitrogen contributes to habitat degradation, algal blooms, toxicity, hypoxia (reduced dissolved oxygen), anoxia (absence of dissolved oxygen), reduction of sea grass habitats, fish kills, and decrease in biodiversity.[270,271,272,273,274,275] Each of these potential impacts carries ecological and economic consequences. Ecosystem services provided by estuaries include fish and shellfish harvest, waste assimilation, and recreational activities.[276]

Increased nitrogen deposition can cause shifts in community composition, reduced hypolimnetic DO, reduced biodiversity, and mortality of submerged aquatic vegetation. The form of deposited nitrogen can significantly affect phytoplankton community composition in estuarine and marine environments. Small diatoms are more efficient in using nitrate than NH_4^+. Increasing NH_4^+ deposition relative to nitrate in the eastern U.S. favors small diatoms at the expense of large diatoms. This alters the foundation of the food web. Submerged aquatic vegetation is important to the quality of estuarine ecosystem habitats because it provides habitat for a variety of aquatic organisms, absorbs excess nutrients, and traps sediments. Nutrient enrichment is the major driving factor contributing to declines in submerged aquatic vegetation coverage. The Mid-Atlantic region is the most heavily impacted area in terms of moderate or high loss of submerged aquatic vegetation due to eutrophication.

Estuarine and Coastal Aquatic

Estuaries and coastal waters tend to be nitrogen-limited and are therefore inherently sensitive to increased atmospheric nitrogen loading.[277,278] The U.S. national estuary condition assessment completed in 2007[279] found that the most impacted estuaries in the U.S. occurred in the mid- Atlantic region and the estuaries with the lowest symptoms of eutrophication were in the North Atlantic. Nitrogen nutrient enrichment is a major environmental problem for coastal regions of the U.S., especially in the eastern and Gulf Coast regions. Of 138 estuaries examined in the national estuary assessment, 44 were identified as showing symptoms of nutrient over-enrichment. Estuaries are among the most biologically productive ecosystems on Earth and provide critical habitat for an enormous diversity of life forms, especially fish. Of the 23 estuaries examined in the national

assessment in the Northeast, 61% were classified as moderately to severely degraded.[280] Other regions had mixtures of low, moderate, and high degree of eutrophication (See Figure 3.3-11).

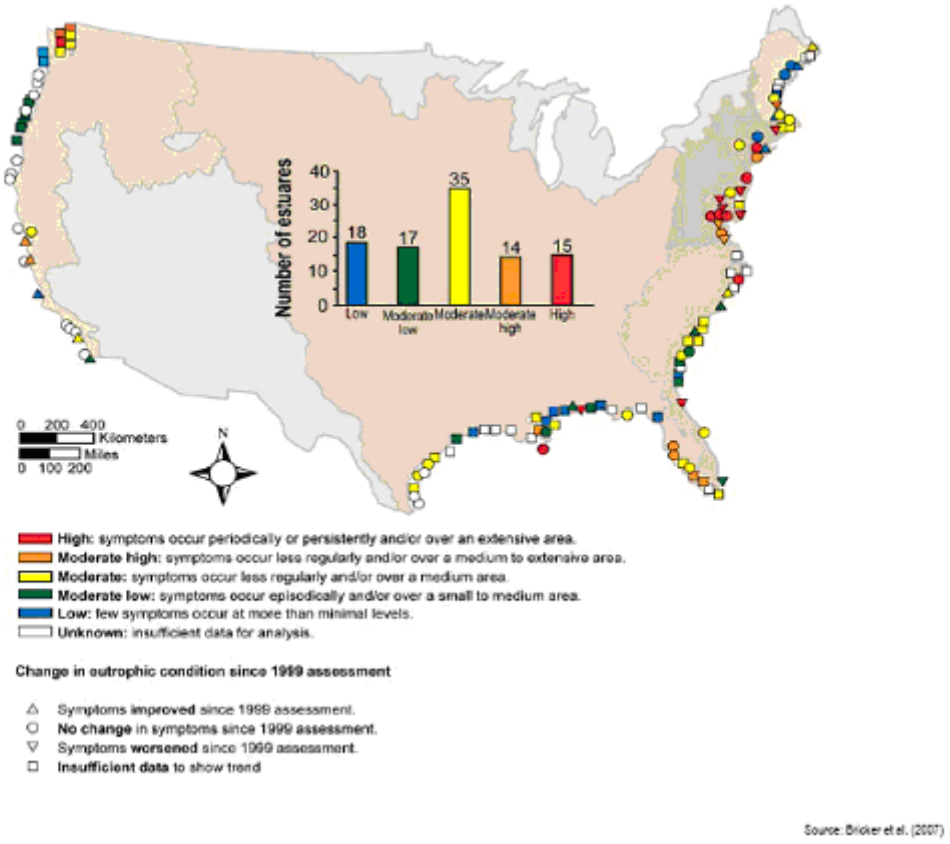


Figure 3.3-11 Overall Eutrophication Condition on a National Scale

The national assessment also evaluated the future outlook of the nation's estuaries based on population growth and future management plans. They predicted that trophic conditions would worsen in 48 estuaries, stay the same in 11, and improve in only 14 by the year 2020. Between 1999 and 2007, an equal number of estuary systems have improved their trophic status as have worsened. The assessed estuarine surface area with high to moderate/high eutrophic conditions have stayed roughly the same, from 72% in1999,[281] to 78% in the 2007 assessment.[282]

3.3.1.7 Ecological Effects of Acidification

The principal factor governing the sensitivity of terrestrial and aquatic ecosystems to acidification from nitrogen and sulfur deposition is geology (particularly surficial geology).[283] Geologic formations having low base cation supply generally underlie the watersheds of acid-sensitive lakes and streams. Bedrock geology has been used in numerous acidification

studies.[284,285,286,287,288] Other factors contributing to the sensitivity of soils and surface waters to acidifying deposition, include: topography, soil chemistry, land use, and hydrologic flow path.

Terrestrial

Acidifying deposition has altered major biogeochemical processes in the U.S. by increasing the nitrogen and sulfur content of soils, accelerating nitrate and sulfate leaching from soil to drainage waters, depleting base cations (especially calcium and magnesium) from soils, and increasing the mobility of aluminum. Inorganic aluminum is toxic to some tree roots. Plants affected by high levels of aluminum from the soil often have reduced root growth, which restricts the ability of the plant to take up water and nutrients, especially calcium.[289] These direct effects can, in turn, influence the response of these plants to climatic stresses such as droughts and cold temperatures. They can also influence the sensitivity of plants to other stresses, including insect pests and disease[290] leading to increased mortality of canopy trees. In the U.S. terrestrial effects of acidification are best described for forested ecosystems (especially red spruce and sugar maple ecosystems) with additional information on other plant communities, including shrubs and lichen.[291] There are several indicators of stress to terrestrial vegetation including percent dieback of canopy trees, dead tree basal area (as a percent), crown vigor index and fine twig dieback.[292]

Health, Vigor, and Reproduction of Tree Species in Forests

Both coniferous and deciduous forests throughout the eastern U.S. are experiencing gradual losses of base cation nutrients from the soil due to accelerated leaching for acidifying deposition. This change in nutrient availability may reduce the quality of forest nutrition over the long term. Evidence suggests that red spruce and sugar maple in some areas in the eastern U.S. have experienced declining health as a consequence of this deposition. For red spruce, (*picea rubens*) dieback or decline has been observed across high elevation landscapes of the northeastern U.S., and to a lesser extent, the southeastern U.S. Acidifying deposition has been implicated as a causal factor.[293] Since the 1980s, red spruce growth has increased at both the higher- and lower-elevation sites corresponding to a decrease in SO_2 emissions in the U.S. (to about 20 million tons/year by 2000), while NO_X emissions held fairly steady (at about 25 million tons/year). Research indicates that annual emissions of sulfur plus NO_X explained about 43% of the variability in red spruce tree ring growth between 1940 and 1998, while climatic variability accounted for about 8% of the growth variation for that period.[294] The observed dieback in red spruce has been linked, in part, to reduced cold tolerance of the spruce needles, caused by acidifying deposition. Results of controlled exposure studies showed that acidic mist or cloud water reduced the cold tolerance of current-year needles by 3 to 10° F.[295] More recently studies have found a link between availability of soil calcium and winter injury.[296] Figure 3.3-12 shows the distribution of red spruce (brown) and sugar maple (green) in the eastern U.S.

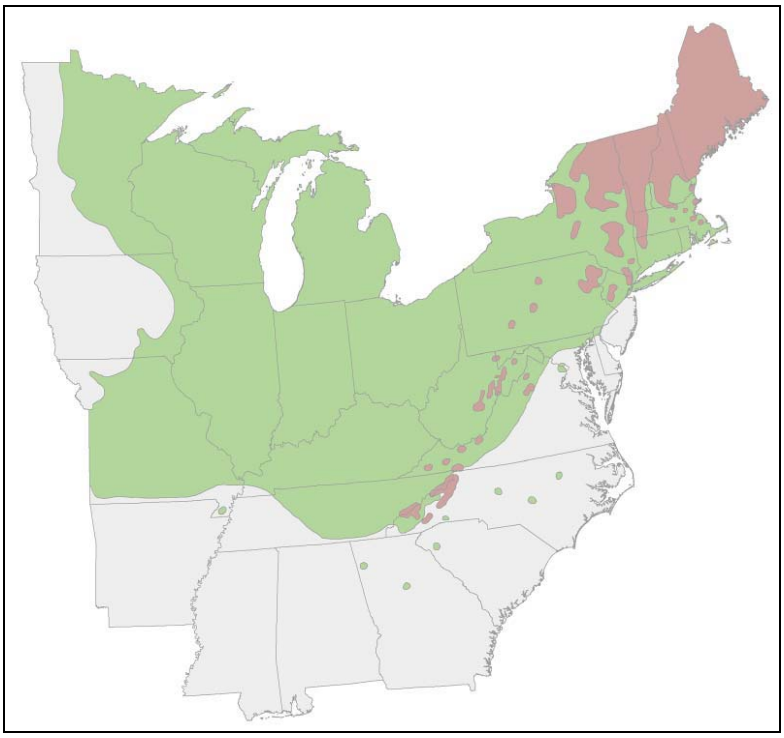


Figure 3.3-12 Distribution of Red Spruce (pink) and Sugar Maple (green) in the Eastern U.S.[297]

In hardwood forests, species nutrient needs, soil conditions, and additional stressors work together to determine sensitivity to acidifying deposition. Stand age and successional stage also can affect the susceptibility of hardwood forests to acidification effects. In northeastern hardwood forests, older stands exhibit greater potential for calcium depletion in response to acidifying deposition than younger stands. Thus, with the successional change from pin cherry (*Prunus pensylvanica*), striped maple (*Acer pensylvanicum*), white ash (*Fraxinus americana*), yellow birch and white birch (*Betula papyrifera*) in younger stands to beech and red maple in older stands, there is an increase in sensitivity to acidification.[298]

Sugar maple (Acer saccharum) is the deciduous tree species of the northeastern U.S. and central Appalachian Mountain region (See Figure 3-14) that is most commonly associated with adverse acidification-related effects of nitrogen and sulfur deposition.[299] In general, evidence indicates that acidifying deposition in combination with other stressors is a likely contributor to the decline of sugar maple trees that occur at higher elevation, on geologies dominated by sandstone or other base-poor substrate, and that have base-poor soils having high percentages of rock fragments.[300]

Loss of calcium ions in the base cations has also been implicated in increased susceptibility of flowering dogwood (*Cornus florida*) to its most destructive disease, dogwood anthracnose- a mostly fatal disease. Figure 3.3-13 shows the native range of flowering dogwood in the U.S. (dark gray) as well as the range of the anthracnose disease as of 2002 in the eastern U.S. (red). Flowering dogwood is a dominant understory species of hardwood forests in the eastern U.S.[301]

Source: Holzmueller et al. (2006). Reprinted with permission.

Figure 3.3-13 Native Range of Flowering Dogwood (dk gray) and the Documented Range of Dogwood Anthracnose (red) Source: Holzmueller et al (2006)

Limited data exists on the possible effects of nitrogen and sulfur deposition on the acid-based characteristics of forests in the U.S. other than spruce-fire and northern hardwood forests ecosystems as described above. [302]

Health and Biodiversity of Other Plant Communities

Shrubs

Available data suggest that it is likely that a variety of shrub and herbaceous species are sensitive to base cation depletion and/or aluminum toxicity. However, conclusive evidence is generally lacking. [303]

Lichens

Lichens and bryophytes are among the first components of the terrestrial ecosystem to be affected by acidifying deposition.[304] Vulnerability of lichens to increased nitrogen input is generally greater than that of vascular plants.[305] Even in the Pacific Northwest, which receives uniformly low levels of nitrogen deposition, changes from acid-sensitive and nitrogen-sensitive to pollution tolerant nitrophillic lichen taxa are occurring in some areas.[306] Lichens remaining in areas affected by acidifying deposition were found to contain almost exclusively the families Candelariaccae, Physciaceae, and Teloschistaceae.[307]

Effects of sulfur dioxide exposure to lichens includes: reduced photosynthesis and respiration, damage to the algal component of the lichen, leakage of electrolytes, inhibition of nitrogen fixation, reduced K absorption, and structural changes.[308] Additional research has concluded that the sulfur:nitrogen exposure ratio is as important as pH in causing toxic effects on lichens. Thus, it is not clear to what extent acidity may be the principal stressor under high levels of air pollution exposure. The toxicity of sulfur dioxide to several lichen species is

greater under acidic conditions than under neutral conditions.[309] The effects of excess nitrogen deposition to lichen communities are discussed in Section 3.3.1.5.

Artic and Alpine Tundra

The possible effects of acidifying deposition on arctic and alpine plant communities are also of concern to the U.S.[310] Especially important in this regard is the role of nitrogen deposition in regulating ecosystem nitrogen supply and plant species composition. Soil acidification and base cation depletion in response to acidifying deposition have not been documented in arctic or alpine terrestrial ecosystems in the U.S. Such ecosystems are rare and spatially limited in the eastern U.S., where acidifying deposition levels have been high. These ecosystems are more widely distributed in the western U.S. and throughout much of Alaska, but acidifying deposition levels are generally low in these areas. Key concerns are for listed threatened or endangered species and species diversity.

Aquatic Ecosystems

Aquatic effects of acidification have been well studied in the U.S. and elsewhere at various trophic levels. These studies indicate that aquatic biota have been affected by acidification at virtually all levels of the food web in acid sensitive aquatic ecosystems. Effects have been most clearly documented for fish, aquatic insects, other invertebrates, and algae.

Biological effects are primarily attributable to a combination of low pH and high inorganic aluminum concentrations. Such conditions occur more frequently during rainfall and snowmelt that cause high flows of water and less commonly during low-flow conditions, except where chronic acidity conditions are severe. Biological effects of episodes include reduced fish condition factor, changes in species composition and declines in aquatic species richness across multiple taxa, ecosystems and regions. These conditions may also result in direct mortality.[311] Biological effects in aquatic ecosystems can be divided into two major categories: effects on health, vigor, and reproductive success; and effects on biodiversity.

3.3.1.8 Nitrogen and Sulfur Deposition Maps for the U.S – Contribution of International Shipping in 2020 with and without an ECA

Air quality modeling conducted by the U.S. government shows that without any further emission controls, in 2020, shipping activities will contribute to the serious problems of acidification and nutrient enrichment in the U.S by adding significant amounts to nitrogen and sulfur deposition across the U.S. Specifically, in 2020, annual total sulfur deposition attributable to international shipping will range from 10% to more than 25% of total sulfur deposition along the entire Atlantic, Gulf of Mexico, and Pacific coastal areas of the U.S. and this level of impact will extend inland for hundreds of kilometers (See Figure 3.3-14). Of equal significance, international shipping will contribute to total annual sulfur deposition not only along all U.S. coastal areas but throughout the entire U.S. land mass, impacting sensitive terrestrial and aquatic ecosystems in the vast interior and heartland regions of the U.S.

Contributions to sulfur deposition will range from 1% to 5% in ecosystems located throughout the interior sections of the U.S.

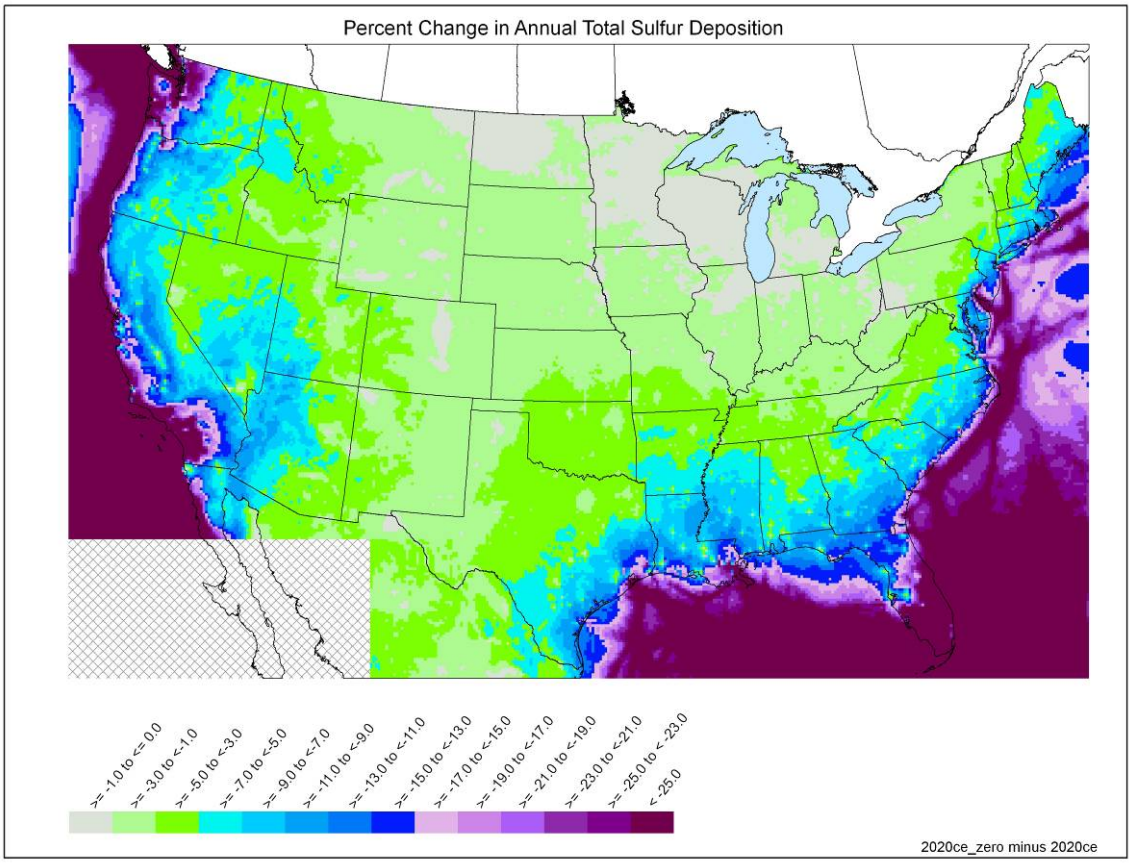


Figure 3.3-14 Percent Contribution in 2020 of Ships to Annual Total Sulfur Deposition in the U.S.

With respect to nitrogen deposition, in 2020, annual total nitrogen deposition from international shipping will range from about 9% to more than 25% along the entire U.S. Atlantic, Pacific and Gulf of Mexico coastal areas. Nitrogen deposition from international shipping will also extend inland for hundreds of kilometers. In addition, throughout the remaining land areas of the U.S., international shipping will also contribute to annual total nitrogen deposition--in the range of 1% to 5% by 2020 (See Figure 3.3-15).

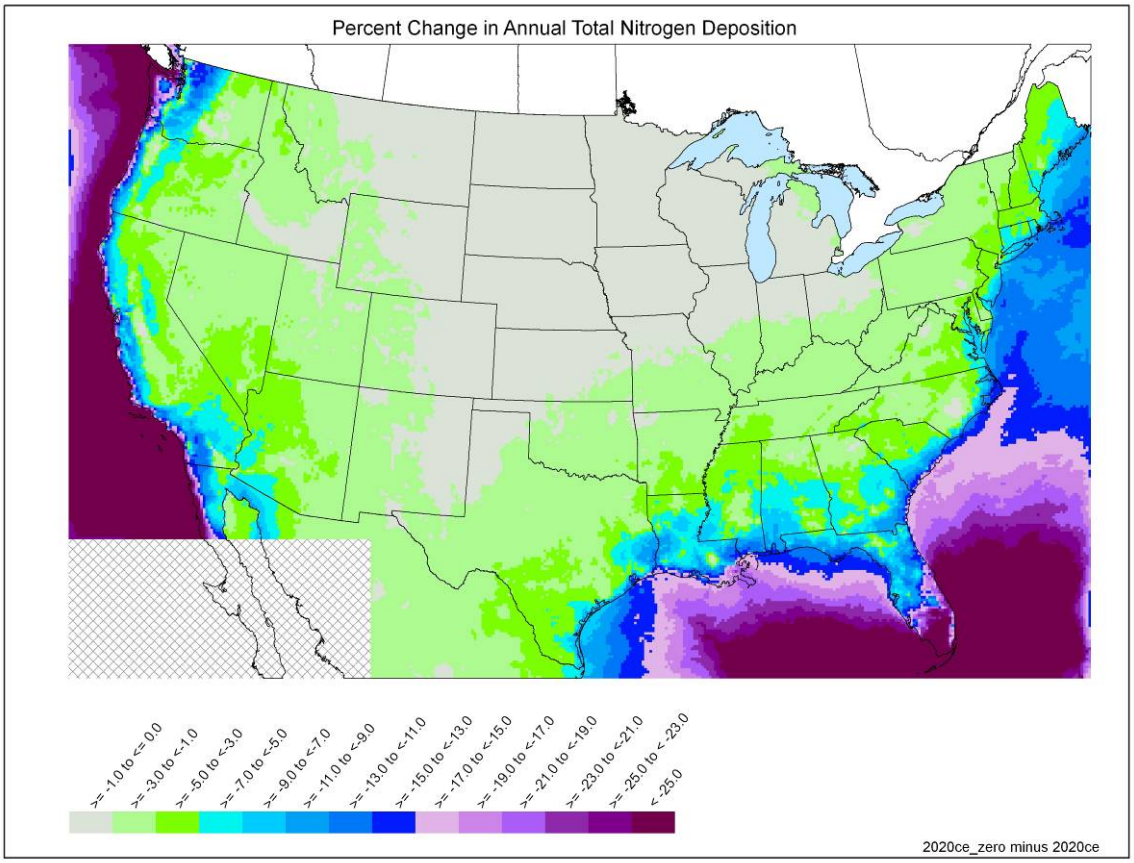


Figure 3.3-15 Percent Contribution in 2020 of Ships to Annual Total Nitrogen Deposition in the U.S.

If the proposed ECA were adopted, reductions in nitrogen deposition would result by 2020, benefiting many sensitive ecological areas throughout the U.S. Areas benefiting are described in detail in section 3.3.1.1 and include sensitive forests, wetlands such as freshwater bogs and marshes, lakes and streams throughout the entire U.S. Figure 3.3-16 illustrates the nitrogen deposition reductions that would occur along U.S. coastlines in 2020 as well as reductions occurring within the interior of the U.S. Reductions would range from 3% to 7% along the entire Atlantic and Gulf Coasts with a few regions, such as southern Louisiana and Florida, experiencing nitrogen reductions up to 9%. Along the Pacific Coast, modeling shows that nitrogen deposition reductions would be higher, ranging from 3% to 15% on land and as high as 20% in some coastal waters.

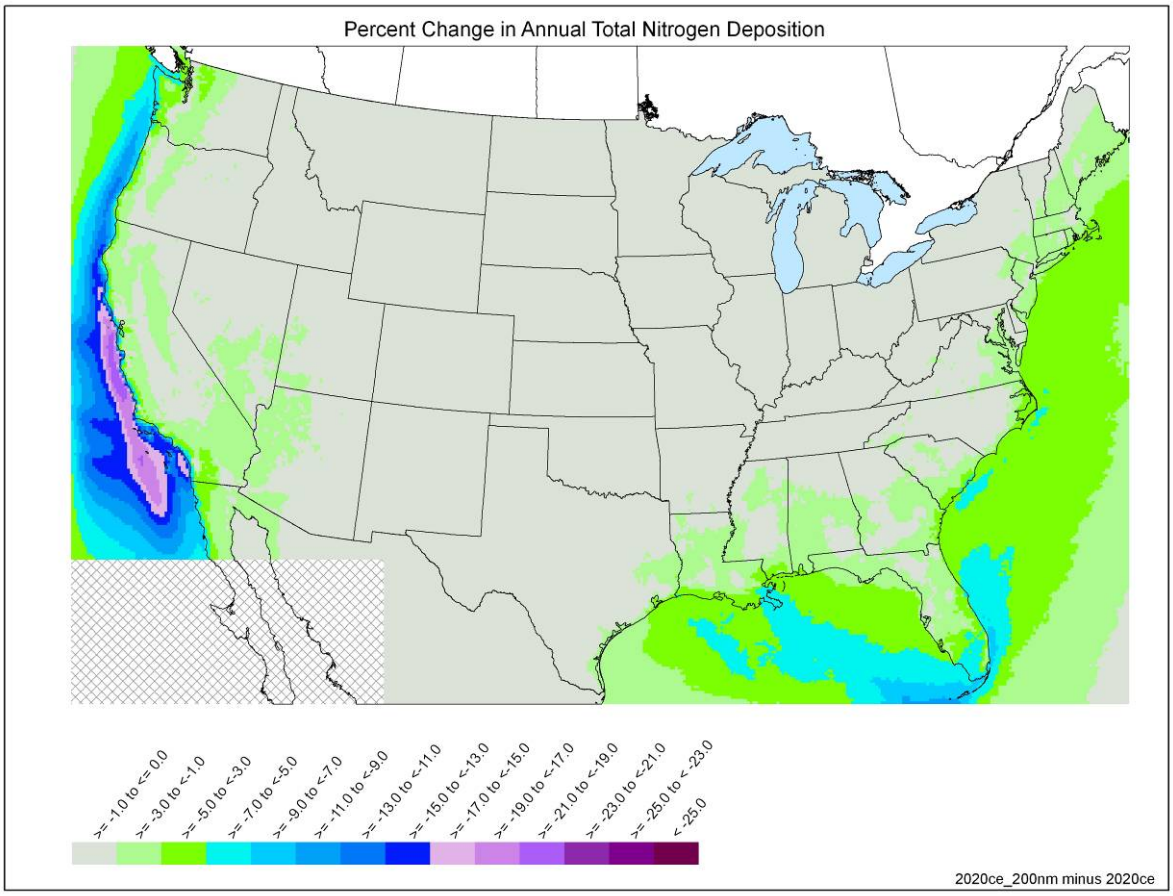


Figure 3.3-16 Percent Change in Annual Total Nitrogen over the U.S. Modeling Domain for the ECA Modeling Scenario.

With respect to sulfur deposition, adopting the proposed ECA would result in reducing sulfur deposition levels in 2020; in some regions by more than 25%. Figure 3.3-17 illustrates the sulfur deposition reductions occurring throughout the U.S. In some individual U.S. watersheds, consisting of offshore islands or close to coastal areas, sulfur deposition levels would be reduced by up to 80%. More generally, the Northeast Atlantic Coastal region would experience sulfur deposition reductions from C3 vessels ranging from 7% to 25% while the Southeast Atlantic Coastal region would experience reductions ranging from 7% to more than 25%. Sulfur deposition would be reduced in the Gulf Coast region from 3% to more than 25%. Along the West Coast of the U.S. sulfur deposition reductions exceeding 25% would occur in the entire Los Angeles Basin in the State of California. The Pacific Northwest would also see significant sulfur deposition reductions ranging from 4% to more than 25%. As importantly, sulfur reductions due to the proposed ECA would also impact the entire U.S. land mass with even interior sections of the U.S. experiencing reductions of 1%. Together, these reductions would assist the U.S. in its efforts to reduce acidification impacts associated with nitrogen and sulfur deposition in both terrestrial and aquatic ecosystems in coastal areas of the U.S. as well as within the interior of the U.S.

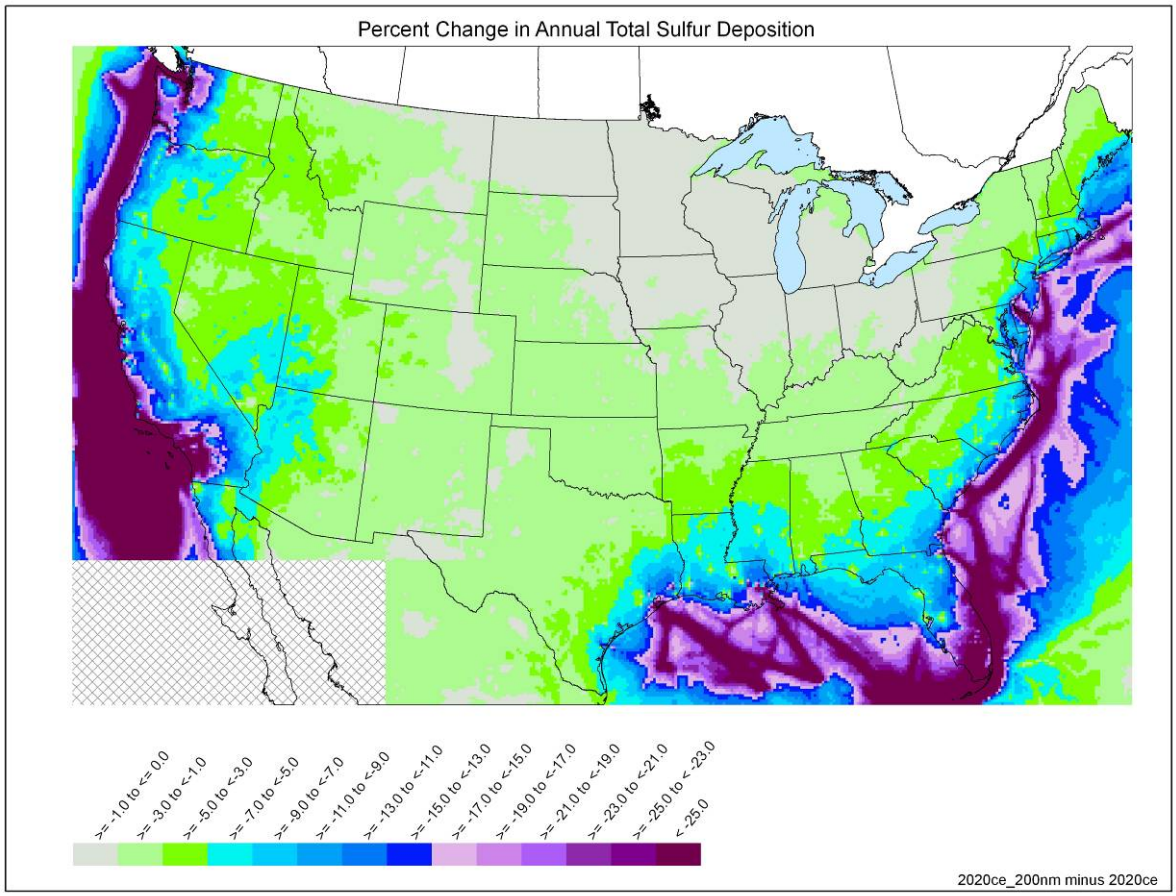


Figure 3.3-17 Percent Change in Annual Total Sulfur over the U.S. Modeling Domain for the ECA Modeling Scenario.

Appendix 3B presents both the range as well as the average total nitrogen and total sulfur deposition changes in 2020 for CMAQ modeling scenarios over 18 specific U.S. subregions. In the case of the proposed ECA, sulfur deposition levels were reduced by on average from 0 to 19 percent over these large drainage regions. In individual HUCs consisting of offshore islands or close to coastal areas, sulfur deposition levels in 2020 were improved by as much as 78% in the proposed ECA while nitrogen deposition levels were improved by as much as 13% in some coastal areas.

3.3.1.8.1 Methodology

The CMAQ model provides estimates of the amount of nitrogen and sulfur deposition in each of the simulated scenarios. The modeling indicated that the shipping sector contributes to acid deposition over the U.S. modeling domain and that these impacts will grow by 2020, if no control measures are adopted by then. Figures 3-16 and 3-17 show the percent change in total nitrogen and total sulfur deposition in 2020 expected to result from the application of the proposed ECA. These plots are based on absolute outputs from the CMAQ modeling.

Additionally, we conducted additional analyses using a separate methodology in which the CMAQ outputs were used to estimate the impacts on deposition levels in a manner similar to how the model is used for ozone and fine particulate matter. In this methodology, CMAQ outputs of annual wet deposition from the 2002 base year model run are used in conjunction with annual wet deposition predictions from the control or future case scenarios to calculate relative reduction factors (RRFs) for wet deposition. Separate wet deposition RRFs are calculated for reduced nitrogen, oxidized nitrogen, and sulfur. These RRFs are multiplied by the corresponding measured annual wet deposition of reduced nitrogen, oxidized nitrogen, and sulfur from the National Atmospheric Deposition Program (NADP) network. The result will be a projection of the NADP wet deposition for the control or future case scenarios. The projected wet deposition for each of the three species is added to the CMAQ-predicted dry deposition for each of these species to produce total reduced nitrogen, total oxidized nitrogen, and total sulfur deposition for the control/future case scenario. The reduced and oxidized nitrogen depositions are summed to calculate total nitrogen deposition.

This analysis was completed for each individual 8-digit hydrological unit code (HUC) within the U.S. modeling domain. Each 8-digit HUC represents a local drainage basin. There were 2,108 8-digit HUCs considered as part of this analysis. This assessment corroborated the absolute deposition modeling results. Appendix 3B shows the average total nitrogen and total sulfur deposition changes for three CMAQ modeling scenarios over 18 specific subregions. In the case of an ECA adoption, sulfur deposition levels were reduced by 0 to 19 percent over these large drainage regions. In individual HUCs consisting of offshore islands or close to coastal areas, sulfur deposition levels were improved by as much as 78% in the ECA case. Nitrogen deposition levels were improved by as much as 13% in some coastal areas.

3.3.1.9 Case Study: Critical Load Modeling in the Adirondack Mountains of New York State and the Blue Ridge Mountains in the State of Virginia

The Adirondack Mountains of New York and the Blue Ridge Mountains of Virginia have long been a locus for awareness of the environmental issues related to acidifying deposition. Soils and water bodies, such as lakes and streams, usually buffer the acidity from natural rain with "bases," the opposite of acids from the environment. The poor buffering capability of the soils in both these regions make the lakes and streams particularly susceptible to acidification from anthropogenic nitrogen and sulfur atmospheric deposition resulting from nitrogen and sulfur oxides emissions. Consequently, acidic deposition has affected hundreds of lakes and thousands of miles of headwater streams in both of these regions. The diversity of life in these acidic waters has been reduced as a result of acidic deposition.

The critical load approach provides a quantitative estimate of the exposure to one or more pollutants below which significant harmful effects on specific sensitive elements of the environment do not occur according to present knowledge. The critical load for a lake or stream provides a means to gauge the extent to which a water body has recovered from past acid deposition, or is potentially at risk due to current deposition levels. Acid neutralizing capacity (ANC) is an excellent indicator of the health of aquatic organisms such as fish, insects, and invertebrates.

Figure 3.3-18 Locations of lakes and streams used in this assessment

In this case study, the focus is on the combined load of nitrogen and sulfur and deposition below which the ANC level would still support healthy aquatic ecosystems. Critical loads were calculated for 169 lakes in the Adirondack region and 60 streams in Virginia (Figure 3.3-18). The Steady-State Water Chemistry (SSWC) model was used to calculate the critical load, relying on water chemistry data from the USEPA Temporal Intergraded Monitoring of Ecosystems (TIME) and Long-term Monitoring (LTM) programs and model assumptions well supported by the scientific literature. Research studies have shown that surface water with ANC values greater than 50 micro-equivalents per Liter (μeq/L) tend to protect most fish (i.e., brook trout, others) and other aquatic organisms (Table 3.3-4). In this case, the critical load represents the combined deposition load of nitrogen and sulfur to which a lake or stream could be subjected and still have an ANC of 50 μeq/L.

Table 3.3-4 Aquatic Status Categories

CATEGORY LABEL	ANC LEVELS*	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 numbers of individuals in plankton species that are present are greatly reduced.
Severe Concern	0 – 20 μeq/L	Highly sensitive to episodic acidification. During episodes of high acid 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 (more than half of expected species are missing). On average, brook trout populations experience sub-lethal effects, including loss of health and reproduction (fitness). Diversity and distribution of zooplankton communities also decline.
Moderate Concern	50 – 100 μeq/L	Fish species richness begins to decline (sensitive species are lost from lakes). Brook trout populations are sensitive and variable, with possible sub-lethal effects. Diversity and distribution of zooplankton communities begin to decline as species that are sensitive to acid 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 range.

When the critical load is "exceeded," it means that the amount of combined nitrogen and sulfur atmospheric deposition is greater than the critical load for a particular lake or stream, preventing the water body from reaching or maintaining an ANC concentration of 50 μeq/L. Critical loads of combined total nitrogen and sulfur are expressed in terms of ionic charge balance as milliequivalent per square meter per year ($meq/m^2/yr$). Exceedances were calculated from deposition for years 2002 and 2020 with and without emissions from shipping. In year 2002, there was no difference in the percent of lakes or streams in both regions that exceeded the critical load for the case with and without ship emissions (Table 3.3-5). For the year 2020, when ship emissions are present, 33% of lakes in the Adirondack Mountains and 52% of streams in the Virginia Blue Ridge Mountains received greater acid deposition than could be neutralized. When ship emissions were removed from the modeling domain for the year 2020, 31 and 50 percent of lakes and streams, respectively, received greater acid deposition than could be neutralized- a 2% improvement.

Regional Assessment

A regional estimate of the benefits of the reduction in international shipping emissions in 2020 can be derived from scaling up the results from 169 lakes to a larger population of lakes in the Adirondack Mountains. One hundred fifteen lakes of the 169 lakes modeled for critical loads are part of a subset of 1,842 lakes in the Adirondacks, which include all lakes from 0.5 to 2000 ha in size and at least 0.5 meters in depth. Using weighting factors derived from the EMAP probability survey and the critical load calculations from the 115 lakes,

exceedance estimates were derived for the entire 1,842 lakes in the Adirondacks. Based on this approach, 66 fewer lakes in the Adirondack Mountains are predicted to receive nitrogen and sulfur deposition loads below the critical load and would be protected as a result of removing international shipping emissions in 2020.

Currently, no probability survey has been completed for the study area in Virginia. However, the 60 trout streams modeled are characteristic of first and second order streams on non-limestone bedrock in the Blue Ridge Mountains of Virginia. Because 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. In addition, the 60 streams are a subset of 344 streams sampled by the Virginia Trout Stream Sensitivity Study, which can be applied to a population of 304 out of the original 344 streams. Using the 304 streams to which the analysis applies directly as the total, 6 additional streams in this group would be protected as a result of removing international shipping emissions in 2020. However, it is likely that many more of the ~12,000 trout streams in Virginia would benefit from reduced international shipping emissions given the extent of similar bedrock geology outside the study area.

Table 3.3-5 Percent of Modeled Lakes that Exceed the Critical Load for Years 2002 and 2020 with and without International Shipping Emissions. "Zero" Indicates without International Shipping Emissions

	2002	2002 ZERO	2020	2020 ZERO
Adirondack Mountains				
Exceeded Critical Load (%. Lakes)	45	45	33	31
Non-Exceeded Critical Load (%. Lakes)	55	55	73	71
Virginia Blue Ridge Mountains				
Exceeded Critical Load (%. Lakes)	82	82	52	50
Non-Exceeded Critical Load (%. Lakes)	18	18	48	50

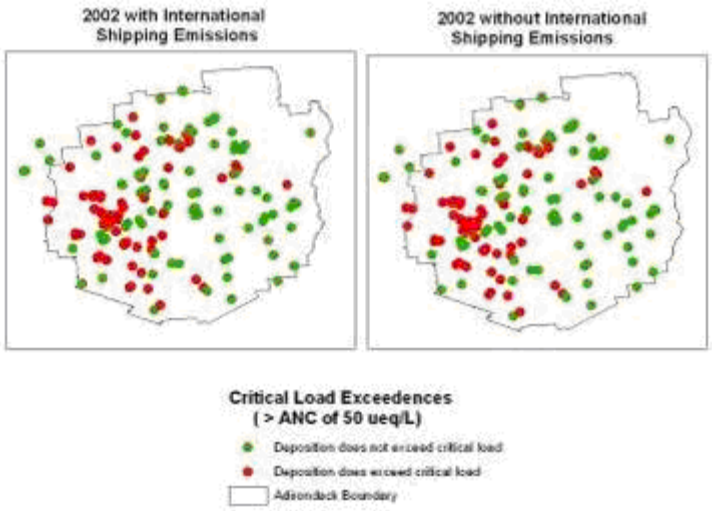


Figure 3.3-19 a. 2002

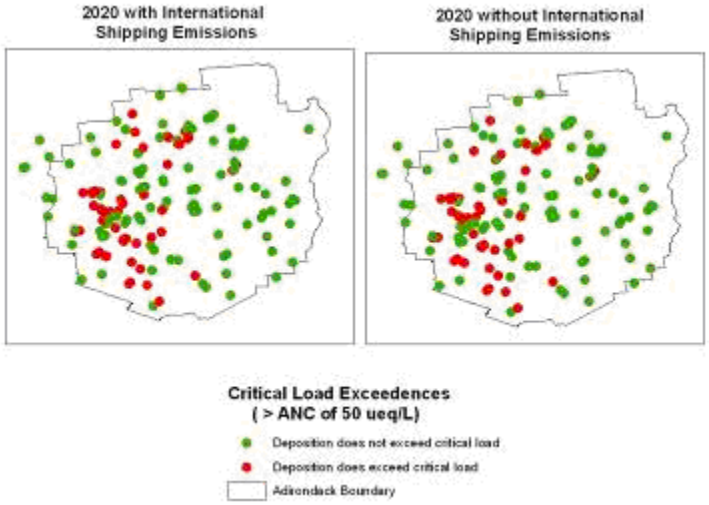


Figure 3.3-19 b. 2020; Critical Load Exceedance for ANC Concentration of 50 µeq/L. Green dots represent lakes in the Adirondack Mountains where current nitrogen and sulfur deposition is below their critical load and maintains an ANC concentration of 50 µeq/L. Red dots are lakes where current nitrogen and sulfur deposition exceeds their limit and the biota are likely impacted

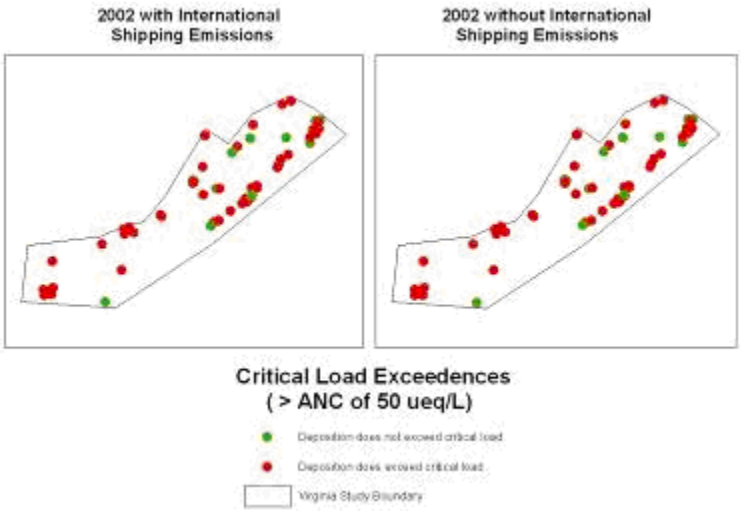


Figure 3.3-20 a. 2002

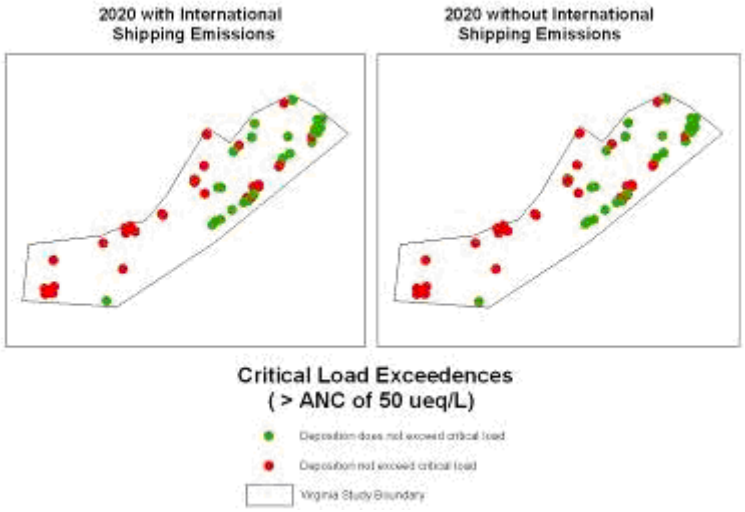


Figure 3.3-20 b . 2020; Critical Load Exceedances for ANC Concentration of 50 µeq/L. Green dots represent streams in the Virginia Blue Ridge Mountains where current nitrogen and sulfur deposition is below their critical load and maintains an ANC concentration of 50 µeq/L. Red dots are streams where current nitrogen and sulfur deposition exceeds their limit and the biota are likely impacted.

3.3.2 Ozone Impacts on Plants and Ecosystems (overview)

There are a number of environmental or public welfare effects associated with the presence of ozone in the ambient air.[312] In this section we discuss the impact of ozone on plants, including trees, agronomic crops and urban ornamentals.

The Air Quality Criteria Document for Ozone and related Photochemical Oxidants notes that "ozone affects vegetation throughout the United States, impairing crops, native vegetation, and ecosystems more than any other air pollutant".[313] Like carbon dioxide (CO_2) and other gaseous substances, ozone enters plant tissues primarily through apertures (stomata) in leaves in a process called "uptake".[314] Once sufficient levels of ozone, a highly reactive substance, (or its reaction products) reaches the interior of plant cells, it can inhibit or damage

essential cellular components and functions, including enzyme activities, lipids, and cellular membranes, disrupting the plant's osmotic (i.e., water) balance and energy utilization patterns.[315,316] If enough tissue becomes damaged from these effects, a plant's capacity to fix carbon to form carbohydrates, which are the primary form of energy used by plants is reduced,[317] while plant respiration increases. With fewer resources available, the plant reallocates existing resources away from root growth and storage, above ground growth or yield, and reproductive processes, toward leaf repair and maintenance, leading to reduced growth and/or reproduction. Studies have shown that plants stressed in these ways may exhibit a general loss of vigor, which can lead to secondary impacts that modify plants' responses to other environmental factors. Specifically, plants may become more sensitive to other air pollutants, more susceptible to disease, insect attack, harsh weather (e.g., drought, frost) and other environmental stresses. Furthermore, there is evidence that ozone can interfere with the formation of mycorrhiza, essential symbiotic fungi associated with the roots of most terrestrial plants, by reducing the amount of carbon available for transfer from the host to the symbiont.[318,319]

This ozone damage may or may not be accompanied by visible injury on leaves, and likewise, visible foliar injury may or may not be a symptom of the other types of plant damage described above. When visible injury is present, it is commonly manifested as chlorotic or necrotic spots, and/or increased leaf senescence (accelerated leaf aging). Because ozone damage can consist of visible injury to leaves, it can also reduce the aesthetic value of ornamental vegetation and trees in urban landscapes, and negatively affects scenic vistas in protected natural areas.

Ozone can produce both acute and chronic injury in sensitive species depending on the concentration level and the duration of the exposure. Ozone effects also tend to accumulate over the growing season of the plant, so that even lower concentrations experienced for a longer duration have the potential to create chronic stress on sensitive vegetation. Not all plants, however, are equally sensitive to ozone. Much of the variation in sensitivity between individual plants or whole species is related to the plant's ability to regulate the extent of gas exchange via leaf stomata (e.g., avoidance of ozone uptake through closure of stomata)[320,321,322] Other resistance mechanisms may involve the intercellular production of detoxifying substances. Several biochemical substances capable of detoxifying ozone have been reported to occur in plants, including the antioxidants ascorbate and glutathione. After injuries have occurred, plants may be capable of repairing the damage to a limited extent.[323]

Because of the differing sensitivities among plants to ozone, ozone pollution can also exert a selective pressure that leads to changes in plant community composition. Given the range of plant sensitivities and the fact that numerous other environmental factors modify plant uptake and response to ozone, it is not possible to identify threshold values above which ozone is consistently toxic for all plants. The next few paragraphs present additional information on ozone damage to trees, ecosystems, agronomic crops and urban ornamentals.

Ozone also has been conclusively shown to cause discernible injury to forest trees.[324,325] In terms of forest productivity and ecosystem diversity, ozone may be the pollutant with the greatest potential for regional-scale forest impacts. Studies have

demonstrated repeatedly that ozone concentrations commonly observed in polluted areas can have substantial impacts on plant function.[326,327]

Because plants are at the base of the food web in many ecosystems, changes to the plant community can affect associated organisms and ecosystems (including the suitability of habitats that support threatened or endangered species and below ground organisms living in the root zone). Ozone impacts at the community and ecosystem level vary widely depending upon numerous factors, including concentration and temporal variation of tropospheric ozone, species composition, soil properties and climatic factors.[328] In most instances, responses to chronic or recurrent exposure in forested ecosystems are subtle and not observable for many years. These injuries can cause stand-level forest decline in sensitive ecosystems.[329,330,331] It is not yet possible to predict ecosystem responses to ozone with much certainty; however, considerable knowledge of potential ecosystem responses has been acquired through long-term observations in highly damaged forests in the United States.

Laboratory and field experiments have also shown reductions in yields for agronomic crops exposed to ozone, including vegetables (e.g., lettuce) and field crops (e.g., cotton and wheat). The most extensive field experiments, conducted under the National Crop Loss Assessment Network (NCLAN) examined 15 species and numerous cultivars. The NCLAN results show that "several economically important crop species are sensitive to ozone levels typical of those found in the United States."[332] In addition, economic studies have shown reduced economic benefits as a result of predicted reductions in crop yields associated with observed ozone levels.[333,334,335]

Urban ornamentals represent an additional vegetation category likely to experience some degree of negative effects associated with exposure to ambient ozone levels. It is estimated that more than $20 billion (1990 dollars) are spent annually on landscaping using ornamentals, both by private property owners/tenants and by governmental units responsible for public areas.[336] This is therefore a potentially costly environmental effect. However, in the absence of adequate exposure-response functions and economic damage functions for the potential range of effects relevant to these types of vegetation, no direct quantitative analysis has been conducted.

Air pollution can have noteworthy cumulative impacts on forested ecosystems by affecting regeneration, productivity, and species composition.[337] In the U.S., ozone in the lower atmosphere is one of the pollutants of primary concern. Ozone injury to forest plants can be diagnosed by examination of plant leaves. Foliar injury is usually the first visible sign of injury to plants from ozone exposure and indicates impaired physiological processes in the leaves.[338]

This indicator is based on data from the U.S. Department of Agriculture (USDA) Forest Service Forest Inventory and Analysis (FIA) program. As part of its Phase 3 program, formerly known as Forest Health Monitoring, FIA examines ozone injury to ozone-sensitive plant species at ground monitoring sites in forest land across the country. For this indicator, forest land does not include woodlots and urban trees. Sites are selected using a systematic sampling grid, based on a global sampling design.[339,340] At each site that has at least 30 individual plants of at least three ozone-sensitive species and enough open space to ensure

that sensitive plants are not protected from ozone exposure by the forest canopy, FIA looks for damage on the foliage of ozone-sensitive forest plant species. Because ozone injury is cumulative over the course of the growing season, examinations are conducted in July and August, when ozone injury is typically highest.

Monitoring of ozone injury to plants by the USDA Forest Service has expanded over the last 10 years from monitoring sites in ten states in 1994 to nearly 1,000 monitoring sites in 41 states in 2002. The data underlying this indicator are based on averages of all observations collected in 2002, the latest year for which data are publicly available at the time the study was conducted, and are broken down by EPA Region. Ozone damage to forest plants is classified using a subjective five-category biosite index based on expert opinion, but designed to be equivalent from site to site. Ranges of biosite values translate to no injury, low or moderate foliar injury (visible foliar injury to highly sensitive or moderately sensitive plants, respectively), and high or severe foliar injury, which would be expected to result in tree-level or ecosystem-level responses, respectively.[341, 342]

3.3.2.1 Recent Ozone Impact Data for the U.S.

There is considerable regional variation in ozone-related visible foliar injury to sensitive plants in the U.S. The U.S. EPA has developed an environmental indicator based on data from the U.S. Department of Agriculture (USDA) Forest Service Forest Inventory and Analysis (FIA) program which examines ozone injury to ozone-sensitive plant species at ground monitoring sites in forest land across the country (This indicator does not include woodlots and urban trees). Sites are selected using a systematic sampling grid, based on a global sampling design.[343, 344] Because ozone injury is cumulative over the course of the growing season, examinations are conducted in July and August, when ozone injury is typically highest. The data underlying the indictor in Figure 3.3–21 are based on averages of all observations collected in 2002, the latest year for which data are publicly available at the time the study was conducted, and are broken down by U.S. EPA Regions. Ozone damage to forest plants is classified using a subjective five-category biosite index based on expert opinion, but designed to be equivalent from site to site. Ranges of biosite values translate to no injury, low or moderate foliar injury (visible foliar injury to highly sensitive or moderately sensitive plants, respectively, and high or severe foliar injury, which would be expected to result in tree-level or ecosystem-level responses, respectively.[345]

The highest percentages of observed high and severe foliar injury, those which are most likely to be associated with tree or ecosystem-level responses, are primarily found in the Mid-Atlantic and Southeast regions. In EPA Region 3 (which comprises the States of Pennsylvania, West Virginia, Virginia, Delaware, Maryland and Washington D.C.), 12 percent of ozone-sensitive plants showed signs of high or severe foliar damage, and in Regions 2 (States of New York, New Jersey), and 4 (States of North Carolina, South Carolina, Kentucky, Tennessee, Georgia, Florida, Alabama, and Mississippi) the values were 10 percent and 7 percent, respectively. The sum of high and severe ozone injury ranged from 2 percent to 4 percent in EPA Region 1 (the six New England States), Region 7 (States of Missouri, Iowa, Nebraska and Kansas), and Region 9 (States of California, Nevada, Hawaii and Arizona). The percentage of sites showing some ozone damage was about 45 percent in each of these EPA Regions.

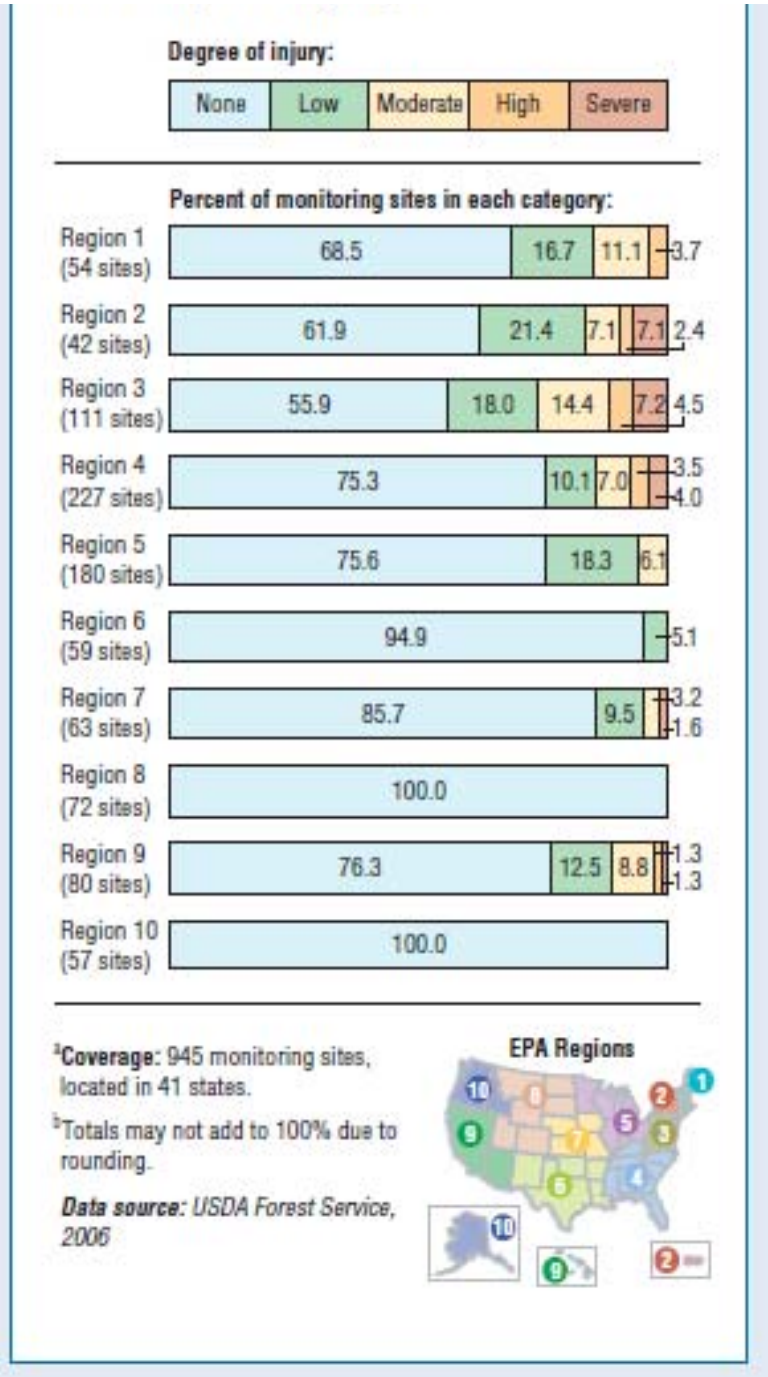


Figure 3.3-21 Ozone Injury to Forest Plants in U.S. by EPA Regions, 2002[ab]

3.3.2.1.1 Indicator Limitations

Field and laboratory studies were reviewed to identify the forest plant species in each region that are highly sensitive to ozone air pollution. Other forest plant species, or even genetic variants of the same species, may not be harmed at ozone levels that cause effects on the selected ozone-sensitive species.

Because species distributions vary regionally, different ozone-sensitive plant species were examined in different parts of the country. These target species could vary with respect to ozone sensitivity, which might account for some of the apparent differences in ozone injury among regions of the U.S.

Ozone damage to foliage is considerably reduced under conditions of low soil moisture, but most of the variability in the index (70 percent) was explained by ozone concentration.[346] Ozone may have other adverse impacts on plants (e.g., reduced productivity) that do not show signs of visible foliar injury.[347]

Though FIA has extensive spatial coverage based on a robust sample design, not all forested areas in the U.S. are monitored for ozone injury. Even though the biosite data have been collected over multiple years, most biosites were not monitored over the entire period, so these data cannot provide more than a baseline for future trends.

3.3.2.1.2 Ozone Impacts on Forest Health

Air pollution can impact the environment and affect ecological systems, leading to changes in the biological community (both in the diversity of species and the health and vigor of individual species). As an example, many studies have shown that ground-level ozone reduces the health of plants including many commercial and ecologically important forest tree species throughout the United States.[348]

When ozone is present in the air, it can enter the leaves of plants, where it can cause significant cellular damage. Since photosynthesis occurs in cells within leaves, the ability of the plant to produce energy by photosynthesis can be compromised if enough damage occurs to these cells. If enough tissue becomes damaged it can reduce carbon fixation and increase plant respiration, leading to reduced growth and/or reproduction in young and mature trees. Ozone stress also increases the susceptibility of plants to disease, insects, fungus, and other environmental stressors (e.g., harsh weather). Because ozone damage can consist of visible injury to leaves, it also reduces the aesthetic value of ornamental vegetation and trees in urban landscapes, and negatively affects scenic vistas in protected natural areas.

Assessing the impact of ground-level ozone on forests in the eastern United States involves understanding the risks to sensitive tree species from ambient ozone concentrations and accounting for the prevalence of those species within the forest. As a way to quantify the risks to particular plants from ground-level ozone, scientists have developed ozone-exposure/tree-response functions by exposing tree seedlings to different ozone levels and measuring reductions in growth as "biomass loss." Typically, seedlings are used because they are easy to manipulate and measure their growth loss from ozone pollution. The mechanisms of susceptibility to ozone within the leaves of seedlings and mature trees are identical, and the decreases predicted using the seedlings should be related to the decrease in overall plant fitness for mature trees, but the magnitude of the effect may be higher or lower depending on the tree species. [349]

Some of the common tree species in the United States that are sensitive to ozone are black cherry (*Prunus serotina*), tulip-poplar (*Liriodendron tulipifera),* eastern white pine

(*Pinus strobus)*. Ozone-exposure/tree-response functions have been developed for each of these tree species, as well as for aspen (*Populus tremuliodes*), and ponderosa pine (*Pinus ponderosa*). Other common tree species, such as oak (*Quercus* spp.*)* and hickory (*Carya* spp.), are not nearly as sensitive to ozone. Consequently, with knowledge of the distribution of sensitive species and the level of ozone at particular locations, it is possible to estimate a "biomass loss" for each species across their range.

3.3.2.2 W126 Modeling and Projected Impact of Ship Emissions on U.S. Forests Biomass

To estimate the biomass loss for the tree species listed above across the eastern United States, the biomass loss for each of the five tree species was calculated using the three-month 12-hour W126 exposure metric at each location and its individual ozone-exposure/tree-response functions. The W126 exposure metric was calculated using monitored data from the AQS air quality monitoring sites. This analysis was done for 2020 with and without international shipping emissions to determine the benefit of lowering shipping emissions on these sensitive tree species in the Eastern half of the U.S.

The biomass loss in the eastern U.S. attributable to international shipping appears to range from 0-6.5 % depending on the particular species. The most sensitive species in the U.S. to ozone-related biomass loss is black cherry; the area of its range with more than 10% biomass loss in 2020 decreased by 8.5% when emissions from ships were removed. Likewise, Table 3-6 indicates that yellow-poplar, eastern white pine, aspen, and ponderosa pine saw areas with more then 2% biomass loss reduced by 2.1% to 3.8% in 2020. The 2% level of biomass loss is important, because a scientific consensus workshop on ozone effects reported that a 2% annual biomass loss causes long term ecological harm due to the potential for compounding effects over multiple years as short-term negative effects on seedlings affect long-term forest health. [350,351] Figure 3.3-22 shows ship emissions' adverse impact on U.S. forest biomass loss in 2020.

Table 3.3-6 The Percent Improvement in Area of the Tree Species Range Between the "Base Case" and "Zero Out" Marine Emissions with Biomass Loss of Greater than 2, 4, 6, and 10% due to Ozone for Year 2020. Units are % Improvement of Area of Species Range.

Tree Species	Percent of Biomass Loss			
	2%	4%	6%	10%
Aspen *Populus tremuloides*	2.4	1.4	0.8	n/a
Black Cherry *Prunus serotina*	n/a	n.c.	2.9	8.5
Ponderosa Pine *Pinus ponderosa*	3.8	2.0	1.5	n/a
Tulip Poplar *Liriodendron tulipifera*	2.1	0.8	n.c.	n/a
E. White Pine *Pinus strobus*	2.8	1.1	0.4	n/a

n.c. - no change in the area
n/a – out of range

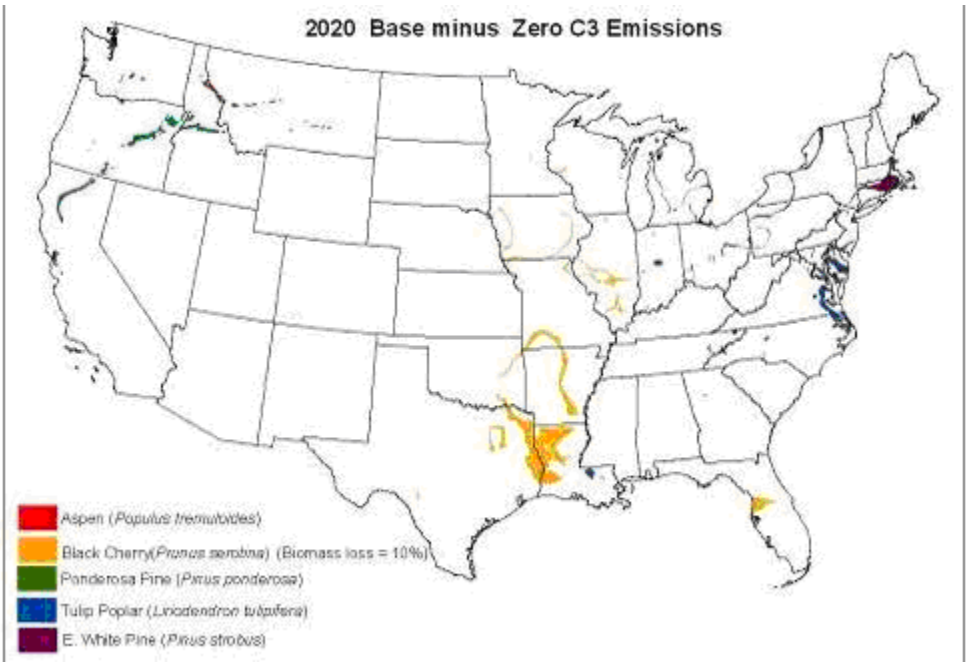


Figure 3.3-22 U.S. Geographic Areas where the Proposed ECA would Reduce Biomass Loss by More than 2%

3.3.2.2.1 Methodology

Outputs from the CMAQ modeling were used to calculate a longer-term ozone exposure metric known as "W126".[352] Previous EPA analyses have concluded that the cumulative, seasonal W126 index is the most appropriate index for relating vegetation response to ambient ozone exposures. The metric is a sigmoidally weighted 3-month sum of all hourly ozone concentrations observed during the daily 12-hr period between 8 am to 8 pm. The three months are the maximum consecutive three months during the ozone season, defined in the ECA modeling as May through September.

As in the ozone and $PM_{2.5}$ modeling, the CMAQ model was used in a relative sense to estimate how ambient W126 levels would change as a result of future growth and/or ECA emissions reductions. The resultant W126 outputs were fed into a separate model which calculated biomass loss from certain tree species as a result of prolonged exposure to ozone. The results of that analysis are discussed below. The CMAQ modeling estimated that ship emissions contributed to high levels of W126 in some coastal areas. This contribution was estimated to range from as much as 30-40 percent in parts of California and Florida. The average contribution from all ship emissions was 8 percent nationally.

3.3.3 Visibility Overview

Emissions from international shipping activity contribute to poor visibility in the U.S. through their primary $PM_{2.5}$ and NO_X emissions (which contribute to the formation of secondary $PM_{2.5}$). These airborne particles degrade visibility by scattering and absorbing

light. Good visibility increases the quality of life where individuals live and work, and where they engage in recreational activities.

Modeling undertaken for the ECA proposal shows that international shipping activities negatively impact visibility by contributing to urban haze in U.S. cities which are located near major deep sea ports and also as regional haze in national parks and wilderness areas throughout the U.S. The U.S. government places special emphasis on protecting visibility in national parks and wilderness areas. Section 169 of the Clean Air Act requires the U.S. government to address existing visibility impairment and future visibility impairment in the 156 national parks exceeding 6,000 acres, and wilderness areas exceeding 5,000 acres, which are categorized as mandatory class I federal areas.

Based on modeling for the ECA proposal, international shipping activities in 2002 contributed to visibility degradation at all of the 133 class I federal areas which have complete Interagency Monitoring of Protected Visual Environments (IMPROVE) ambient data for 2002 or are represented by IMPROVE monitors with complete data.[Z] Absent further emission controls, by 2020, international shipping activities will have an even larger impact on visibility impairment in these class I federal areas. For example, in 2002, approximately 4% of visibility impairment in southern California's Agua Tibia Wilderness was due to shipping activity. U.S. modeling, conducted as part of the ECA proposal, indicates that by 2020 approximately 12.5% of visibility impairment in Agua Tibia will be due to shipping. Likewise, in 2002, 2.7% of visibility impairment in southern Florida's Everglades National Park was due to international shipping, and this will double to 6% by 2020. Even in inland class I federal areas shipping activity is contributing to visibility degradation. In 2020, about 2.5% of visibility degradation in the Grand Canyon National Park, located in the State of Arizona, will be from international shipping, while almost 6% of visibility degradation in the State of Washington's North Cascades National Park will be from shipping emissions.

3.3.3.1 Visibility Monitoring

In conjunction with the U.S. National Park Service, the U.S. Forest Service, other federal land managers, and State organizations in the U.S., the U.S. EPA has supported visibility monitoring in national parks and wilderness areas since 1988. The monitoring network was originally established at 20 sites, but it has now been expanded to 110 sites that represent all but one of the 156 mandatory federal Class I areas across the country. This long-term visibility monitoring network is known as IMPROVE (Interagency Monitoring of PROtected Visual Environments).

IMPROVE provides direct measurement of fine particles that contribute to visibility impairment. The IMPROVE network employs aerosol measurements at all sites, and optical

[Z] There are 156 federally-mandated class I areas which, under the Regional Haze Rule, are required to achieve natural background visibility levels by 2064. These mandatory class I federal areas are mostly national parks, national monuments, and wilderness areas. There are currently 116 IMPROVE monitoring sites (representing all 156 mandatory class I federal areas) collecting ambient $PM_{2.5}$ data at mandatory class I federal areas, but not all of these sites have complete data for 2002.

and scene measurements at some of the sites. Aerosol measurements are taken for PM_{10} and $PM_{2.5}$ mass, and for key constituents of $PM_{2.5}$, such as sulfate, nitrate, organic and elemental carbon, soil dust, and several other elements. Measurements for specific aerosol constituents are used to calculate "reconstructed" aerosol light extinction by multiplying the mass for each constituent by its empirically-derived scattering and/or absorption efficiency, with adjustment for the relative humidity. Knowledge of the main constituents of a site's light extinction "budget" is critical for source apportionment and control strategy development. Optical measurements are used to directly measure light extinction or its components. Such measurements are taken principally with either a transmissometer, which measures total light extinction, or a nephelometer, which measures particle scattering (the largest human-caused component of total extinction). Scene characteristics are typically recorded 3 times daily with 35 millimeter photography and are used to determine the quality of visibility conditions (such as effects on color and contrast) associated with specific levels of light extinction as measured under both direct and aerosol-related methods. Directly measured light extinction is used under the IMPROVE protocol to cross check that the aerosol-derived light extinction levels are reasonable in establishing current visibility conditions. Aerosol-derived light extinction is used to document spatial and temporal trends and to determine how proposed changes in atmospheric constituents would affect future visibility conditions.

Annual average visibility conditions (reflecting light extinction due to both anthropogenic and non-anthropogenic sources) vary regionally across the U.S. The rural East generally has higher levels of impairment than remote sites in the West, with the exception of urban-influenced sites such as San Gorgonio Wilderness (CA) and Point Reyes National Seashore (CA), which have annual average levels comparable to certain sites in the Northeast. Regional differences are illustrated by Figures 4-39a and 4-39b in the CD, which show that, for class I areas, visibility levels on the 20% haziest days in the West are about equal to levels on the 20% best days in the East (CD, p. 4-179).

Higher visibility impairment levels in the East are due to generally higher concentrations of anthropogenic fine particles, particularly sulfates, and higher average relative humidity levels. In fact, sulfates account for 60-86% of the haziness in eastern sites (CD, p. 4-236). Aerosol light extinction due to sulfate on the 20% haziest days is significantly larger in eastern class I areas as compared to western areas (CD, p. 4-182; Figures 4-40a and 4-40b). With the exception of remote sites in the northwestern U.S., visibility is typically worse in the summer months. This is particularly true in the Appalachian region, where average light extinction in the summer exceeds the annual average by 40% (Sisler et al., 1996).

3.3.3.2 Addressing Visibility in the U.S.

The U.S. EPA has two programmatic approaches to address visibility. First, to address the welfare effects of PM on visibility, EPA set secondary $PM_{2.5}$ standards which would act in conjunction with the establishment of a regional haze program. In setting this secondary standard EPA concluded that $PM_{2.5}$ causes adverse effects on visibility in various locations, depending on PM concentrations and factors such as chemical composition and average relative humidity. Second, section 169 of the Clean Air Act provides additional authority to address existing visibility impairment and prevent future visibility impairment in

the 156 national parks, forests and wilderness areas categorized as mandatory class I federal areas (62 FR 38680-81, July 18, 1997).[AA] Figure 3-18 below identifies where each of these parks are located in the U.S. In July 1999 the regional haze rule (64 FR 35714) was put in place to protect the visibility in mandatory class I federal areas. Visibility can be said to be impaired in both $PM_{2.5}$ nonattainment areas and mandatory class I federal areas.[BB] OGVs, powered by Category 3 engines, contribute to visibility concerns in these areas through their primary $PM_{2.5}$ emissions and their NO_X and SO_X emissions which contribute to the formation of secondary $PM_{2.5}$.

Figure 3.3-23 Mandatory Class I Areas in the U.S.

3.3.3.2.1 Current Visibility Impairment

Recently designated $PM_{2.5}$ nonattainment areas indicate that, as of December 2008, over 88 million people live in nonattainment areas for the 1997 $PM_{2.5}$ NAAQS. Thus, at least

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[AA] These areas are defined in section 162 of the Act as those national parks exceeding 6,000 acres, wilderness areas and memorial parks exceeding 5,000 acres, and all international parks which were in existence on August 7, 1977.

[BB] As mentioned above, the EPA has recently proposed to amend the PM NAAQS (71 FR 2620, Jan. 17, 2006). The proposal would set the secondary NAAQS equal to the primary standards for both $PM_{2.5}$ and $PM_{10-2.5}$. EPA also is taking comment on whether to set a separate $PM_{2.5}$ standard, designed to address visibility (principally in urban areas), on potential levels for that standard within a range of 20 to 30 $\mu g/m^3$, and on averaging times for the standard within a range of four to eight daylight hours.

these populations would likely be experiencing visibility impairment, as well as many thousands of individuals who travel to these areas. In addition, while visibility trends have improved in mandatory class I federal areas the most recent data show that these areas continue to suffer from visibility impairment. In eastern parks, average visual range has decreased from 90 miles to 15-25 miles. In the West, visual range has decreased from 140 miles to 35-90 miles. In summary, visibility impairment is experienced throughout the U.S., in multi-state regions, urban areas, and remote mandatory class I federal areas.[353,354] The mandatory federal class I areas are listed in Figure 3.3-23 and in Table 3.3-7.

3.3.3.2.2 Projected Visibility Impairment in U.S. - Impact of Ship Emissions

Based on modeling for the ECA proposal, international shipping activities in 2002 contributed to visibility degradation at all of the 133 class I federal areas which have complete Interagency Monitoring of Protected Visual Environments (IMPROVE) ambient data for 2002 or are represented by IMPROVE monitors with complete data.[CC]. Absent further emission controls, by 2020, international shipping activities will have an even larger impact on visibility deciview levels[DD] in these class I federal areas. The results suggest that controlling emissions from C3 vessels would result in improved visibility deciview levels in all 133 monitored class I federal areas-- although areas would continue to have annual average deciview levels above background in 2020.

The results indicate that reductions in regional haze would occur in all 133 of the areas analyzed as a result of an ECA adoption. The model projects that for all monitored mandatory class I federal areas combined, average visibility on the 20% worst days at these scenic locales would improve by 0.21 deciviews, or 1.2%. The greatest improvements in visibility are in coastal areas. For instance, the Agua Tibia Wilderness area (near Los Angeles) would see 9.4% improvement as a result of the proposed ECA. National parks and national wilderness areas in other parts of the country would also see improvements as a result of ECA controls. For example, the Cape Romain National Wildlife Refuge (South Carolina) would see a 4.6% improvement in visibility; and Acadia National Park (Maine) would see a 4.4% improvement with the proposed ECA. Likewise, in 2002, 2.7% of visibility impairment in southern Florida's Everglades National Park was due to international shipping, and this will double to 6% by 2020. Even in inland class I federal areas international shipping activity is contributing to visibility degradation. In 2020, about 2.5% of visibility degradation in the Grand Canyon National Park located in the state of Arizona will be from international shipping, while almost 6% of visibility degradation in the State of Washington's North

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[CC] There are 156 federally-mandated class I areas which, under the Regional Haze Rule, are required to achieve natural background visibility levels by 2064. These mandatory class I federal areas are mostly national parks, national monuments, and wilderness areas. There are currently 116 IMPROVE monitoring sites (representing all 156 mandatory class I federal areas) collecting ambient PM2.5 data at mandatory class I federal areas, but not all of these sites have complete data for 2002.

[DD] The level of visibility impairment in an area is based on the light-extinction coefficient and a unit less visibility index, called a "deciview", which is used in the valuation of visibility. The deciview metric provides a scale for perceived visual changes over the entire range of conditions, from clear to hazy. Under many scenic conditions, the average person can generally perceive a change of one deciview. The higher the deciview value, the worse the visibility. Thus, an improvement in visibility is a decrease in deciview value.

Cascades National Park will be from international shipping emissions. Table 3.3-7 which follows contains the full visibility results from the 2020 ECA scenario over the 133 analyzed areas.

3.3.3.3 Visibility Modeling

Many scenic areas in the U.S. have reduced visibility because of regional haze. The U.S. EPA is in the midst of a major effort to improve air quality in national parks and wilderness areas, especially for those meteorological situations in which visibility is most degraded. The CMAQ modeling discussed in Section 3.2 was also used to project the impacts of potential ECA-based emissions reductions on visibility conditions over specific national parks and wilderness areas across the U.S. over the 20% worst visibility days at that location.

Table 3.3-7 Visibility Levels in Deciviews for Individual U.S. Class 1 Areas on the 20% Worst Days for Several Scenarios

CLASS 1 AREA (20% WORST DAYS)	STATE	BASELINE VISIBILITY	2020 BASE	ECA	ZERO C3 EMISSIONS	NATURAL BACKGROUND
Sipsey Wilderness	AL	29.03	23.67	23.42	23.32	10.99
Caney Creek Wilderness	AR	26.36	22.20	22.01	21.88	11.58
Upper Buffalo Wilderness	AR	26.27	22.25	22.15	22.11	11.57
Chiricahua NM	AZ	13.43	13.15	13.07	13.00	7.21
Chiricahua Wilderness	AZ	13.43	13.17	13.09	13.02	7.21
Galiuro Wilderness	AZ	13.43	13.18	13.09	13.00	7.21
Grand Canyon NP	AZ	11.66	11.24	11.04	10.96	7.14
Mazatzal Wilderness	AZ	13.35	12.88	12.73	12.61	6.68
Petrified Forest NP	AZ	13.21	12.88	12.76	12.70	6.49
Pine Mountain Wilderness	AZ	13.35	12.74	12.59	12.48	6.68
Saguaro NM	AZ	14.83	14.39	14.31	14.22	6.46
Sierra Ancha Wilderness	AZ	13.67	13.33	13.21	13.10	6.59
Sycamore Canyon Wilderness	AZ	15.25	15.00	14.90	14.84	6.69
Agua Tibia Wilderness	CA	23.50	22.99	20.82	20.11	7.64
Caribou Wilderness	CA	14.15	13.73	13.51	13.43	7.31
Cucamonga Wilderness	CA	19.94	18.34	17.57	17.27	7.06
Desolation Wilderness	CA	12.63	12.29	12.11	12.07	6.12
Dome Land Wilderness	CA	19.43	18.59	18.23	18.14	7.46
Emigrant Wilderness	CA	17.63	17.35	17.14	17.08	7.64
Hoover Wilderness	CA	12.87	12.79	12.68	12.65	7.91
Joshua Tree NM	CA	19.62	17.95	17.30	17.21	7.19
Lassen Volcanic NP	CA	14.15	13.71	13.46	13.37	7.31
Lava Beds NM	CA	15.05	14.47	14.32	14.24	7.86
Mokelumne Wilderness	CA	12.63	12.40	12.21	12.16	6.12
Pinnacles NM	CA	18.46	17.86	17.11	16.89	7.99
Point Reyes NS	CA	22.81	22.38	21.71	21.54	15.77
Redwood NP	CA	18.45	18.26	17.81	17.48	13.91
San Gabriel Wilderness	CA	19.94	17.92	17.12	16.84	7.06
San Gorgonio Wilderness	CA	22.17	20.66	20.45	20.35	7.30
San Jacinto Wilderness	CA	22.17	20.25	19.86	19.55	7.30

CLASS 1 AREA (20% WORST DAYS)	STATE	BASELINE VISIBILITY	2020 BASE	ECA	ZERO C3 EMISSIONS	NATURAL BACKGROUND
South Warner Wilderness	CA	15.05	14.70	14.57	14.51	7.86
Thousand Lakes Wilderness	CA	14.15	13.68	13.42	13.33	7.31
Ventana Wilderness	CA	18.46	18.36	17.72	17.57	7.99
Yosemite NP	CA	17.63	17.32	17.13	17.08	7.64
Black Canyon of the Gunnison NM	CO	10.33	9.77	9.69	9.66	6.24
Eagles Nest Wilderness	CO	9.61	9.05	9.00	8.98	6.54
Flat Tops Wilderness	CO	9.61	9.25	9.20	9.18	6.54
Great Sand Dunes NM	CO	12.78	12.41	12.36	12.34	6.66
La Garita Wilderness	CO	10.33	9.91	9.84	9.81	6.24
Maroon Bells-Snowmass Wilderness	CO	9.61	9.23	9.19	9.16	6.54
Mesa Verde NP	CO	13.03	12.42	12.33	12.28	6.83
Mount Zirkel Wilderness	CO	10.52	10.02	9.99	9.98	6.44
Rawah Wilderness	CO	10.52	10.00	9.97	9.95	6.44
Rocky Mountain NP	CO	13.83	13.09	13.06	13.05	7.24
Weminuche Wilderness	CO	10.33	9.88	9.80	9.77	6.24
West Elk Wilderness	CO	9.61	9.20	9.15	9.12	6.54
Chassahowitzka	FL	26.09	22.37	21.97	21.75	11.21
Everglades NP	FL	22.30	21.75	21.14	20.40	12.15
St. Marks	FL	26.03	22.37	21.96	21.65	11.53
Cohutta Wilderness	GA	30.30	23.29	23.13	23.07	11.14
Okefenokee	GA	27.13	23.86	23.30	23.07	11.44
Wolf Island	GA	27.13	23.76	22.97	22.75	11.44
Craters of the Moon NM	ID	14.00	13.00	12.97	12.94	7.53
Sawtooth Wilderness	ID	13.78	13.66	13.63	13.61	6.43
Mammoth Cave NP	KY	31.37	25.43	25.33	25.30	11.08
Acadia NP	ME	22.89	20.55	19.79	19.62	12.43
Moosehorn	ME	21.72	19.02	18.55	18.38	12.01
Roosevelt Campobello International Park	ME	21.72	19.25	18.58	18.23	12.01
Isle Royale NP	MI	20.74	18.99	18.84	18.81	12.37
Seney	MI	24.16	21.54	21.49	21.47	12.65
Voyageurs NP	MN	19.27	17.55	17.52	17.51	12.06
Hercules-Glades Wilderness	MO	26.75	22.84	22.74	22.72	11.30
Anaconda-Pintler Wilderness	MT	13.41	13.14	13.10	13.07	7.43
Bob Marshall Wilderness	MT	14.48	14.13	14.11	14.09	7.74
Cabinet Mountains Wilderness	MT	14.09	13.55	13.50	13.47	7.53
Gates of the Mountains Wilderness	MT	11.29	10.90	10.87	10.85	6.45
Medicine Lake	MT	17.72	16.20	16.18	16.17	7.90
Mission Mountains Wilderness	MT	14.48	14.02	13.99	13.97	7.74
Scapegoat Wilderness	MT	14.48	14.15	14.12	14.11	7.74

CLASS 1 AREA (20% WORST DAYS)	STATE	BASELINE VISIBILITY	2020 BASE	ECA	ZERO C3 EMISSIONS	NATURAL BACKGROUND
Selway-Bitterroot Wilderness	MT	13.41	13.08	13.02	12.98	7.43
UL Bend	MT	15.14	14.65	14.63	14.62	8.16
Linville Gorge Wilderness	NC	28.77	22.63	22.43	22.34	11.22
Swanquarter	NC	25.49	21.79	21.11	20.99	11.94
Lostwood	ND	19.57	17.45	17.43	17.41	8.00
Theodore Roosevelt NP	ND	17.74	16.44	16.42	16.41	7.79
Great Gulf Wilderness	NH	22.82	19.53	19.34	19.29	11.99
Presidential Range-Dry River Wilderness	NH	22.82	19.53	19.33	19.28	11.99
Brigantine	NJ	29.01	25.27	24.46	24.31	12.24
Bandelier NM	NM	12.22	11.45	11.39	11.36	6.26
Bosque del Apache	NM	13.80	12.93	12.89	12.87	6.73
Gila Wilderness	NM	13.11	12.59	12.52	12.48	6.69
Pecos Wilderness	NM	10.41	10.00	9.93	9.90	6.44
Salt Creek	NM	18.03	16.70	16.66	16.63	6.81
San Pedro Parks Wilderness	NM	10.17	9.52	9.44	9.41	6.08
Wheeler Peak Wilderness	NM	10.41	9.91	9.85	9.82	6.44
White Mountain Wilderness	NM	13.70	12.87	12.82	12.79	6.86
Jarbidge Wilderness	NV	12.07	11.88	11.81	11.78	7.87
Wichita Mountains	OK	23.81	20.45	20.31	20.24	7.53
Crater Lake NP	OR	13.74	13.33	13.20	13.13	7.84
Diamond Peak Wilderness	OR	13.74	13.26	13.11	13.03	7.84
Eagle Cap Wilderness	OR	18.57	17.73	17.69	17.65	8.92
Gearhart Mountain Wilderness	OR	13.74	13.41	13.30	13.25	7.84
Hells Canyon Wilderness	OR	18.55	17.16	17.12	17.07	8.32
Kalmiopsis Wilderness	OR	15.51	15.24	14.85	14.66	9.44
Mount Hood Wilderness	OR	14.86	14.30	13.93	13.64	8.44
Mount Jefferson Wilderness	OR	15.33	14.90	14.62	14.46	8.79
Mount Washington Wilderness	OR	15.33	14.88	14.62	14.46	8.79
Mountain Lakes Wilderness	OR	13.74	13.28	13.14	13.07	7.84
Strawberry Mountain Wilderness	OR	18.57	17.71	17.66	17.62	8.92
Three Sisters Wilderness	OR	15.33	14.93	14.69	14.54	8.79
Cape Romain	SC	26.48	23.51	22.35	22.14	12.12
Badlands NP	SD	17.14	15.63	15.59	15.57	8.06
Wind Cave NP	SD	15.84	14.78	14.75	14.73	7.71
Great Smoky Mountains NP	TN	30.28	24.01	23.81	23.72	11.24
Joyce-Kilmer-Slickrock Wilderness	TN	30.28	23.56	23.35	23.26	11.24

CLASS 1 AREA (20% WORST DAYS)	STATE	BASELINE VISIBILITY	2020 BASE	ECA	ZERO C3 EMISSIONS	NATURAL BACKGROUND
Big Bend NP	TX	17.30	16.25	16.11	16.01	7.16
Carlsbad Caverns NP	TX	17.19	16.05	15.98	15.93	6.68
Guadalupe Mountains NP	TX	17.19	16.03	15.95	15.90	6.68
Arches NP	UT	11.24	10.94	10.86	10.83	6.43
Bryce Canyon NP	UT	11.65	11.41	11.28	11.22	6.86
Canyonlands NP	UT	11.24	10.96	10.90	10.89	6.43
Zion NP	UT	13.24	12.91	12.80	12.73	6.99
James River Face Wilderness	VA	29.12	23.31	23.16	23.12	11.13
Shenandoah NP	VA	29.31	22.77	22.61	22.57	11.35
Lye Brook Wilderness	VT	24.45	21.02	20.77	20.72	11.73
Alpine Lake Wilderness	WA	17.84	16.85	16.56	16.26	8.43
Glacier Peak Wilderness	WA	13.96	13.85	13.53	13.19	8.01
Goat Rocks Wilderness	WA	12.76	12.23	11.95	11.70	8.36
Mount Adams Wilderness	WA	12.76	12.16	11.88	11.67	8.36
Mount Rainier NP	WA	18.24	17.47	17.02	16.66	8.55
North Cascades NP	WA	13.96	13.85	13.46	13.04	8.01
Olympic NP	WA	16.74	16.18	15.87	15.39	8.44
Pasayten Wilderness	WA	15.23	14.89	14.82	14.72	8.26
Dolly Sods Wilderness	WV	29.04	22.46	22.31	22.26	10.39
Otter Creek Wilderness	WV	29.04	22.45	22.30	22.26	10.39
Bridger Wilderness	WY	11.12	10.83	10.78	10.76	6.58
Fitzpatrick Wilderness	WY	11.12	10.87	10.81	10.79	6.58
Grand Teton NP	WY	11.76	11.37	11.32	11.30	6.51
North Absaroka Wilderness	WY	11.45	11.17	11.14	11.13	6.86
Red Rock Lakes	WY	11.76	11.45	11.40	11.38	6.51
Teton Wilderness	WY	11.76	11.43	11.38	11.36	6.51
Washakie Wilderness	WY	11.45	11.19	11.16	11.15	6.86
Yellowstone NP	WY	11.76	11.40	11.35	11.33	6.51

Appendices

Appendix 3A

Once air pollutants have been emitted into the atmosphere, the processes that determine pollutant concentrations in space and time are largely determined by meteorology. This portion of the document describes the relevant meteorological conditions within the proposed areas that contribute to at-sea emissions being transported to populated areas and contributing to harmful human health and ecological impacts.

As noted elsewhere in this document, NO_X, SO_X, and direct particulate matter are emitted from ships. These pollutants and the pollutants that are secondarily formed from these emissions can have atmospheric lifetimes of 5-10 days before being significantly dispersed, deposited, or converted to other species (Clarke et al., 2001; Karamchandani et al., 2006). As a result of these rather long residence times in the atmosphere, it is important to consider similar meteorological scales when determining the potential impacts of ship emissions on human health and ecosystems. Thus, while meteorological phenomena of all sizes affect the eventual impacts of ship emissions, the longer range regional transport of pollutants from shipping is largely dictated by synoptic scale meteorological patterns.

Prevailing wind patterns can vary by season and by location over the United States, but it is common for air masses to have a maritime influence especially looking back at time periods of 5-10 days. Over parts of the U.S., this is readily evident from regional reanalyses of ambient meteorological conditions. Figures 3A-1 and 3A-2 show prevailing winds over the course of last year (2008) based on the NCEP Regional Reanalysis dataset (Mesinger, 2006) which is derived from the Eta weather forecast model as guided by assimilation of large volumes of measured meteorological data. The maps show the monthly mean wind barbs. These wind barbs are comprised of two straight lines, the longest of which indicates the monthly mean wind direction. The shorter line indicates the speed of the monthly mean wind vector. The wind blows from the intersection of the two lines to the end of the longer line. Caution should be exercised when viewing these figures, as there are certainly individual hours and days in which the winds deviate from the monthly means. Additionally, while 2008 was generally a representative year[EE], other years strongly influenced by extreme phases of ocean-atmospheric oscillations, such as the El Nino Southern Oscillation (ENSO) could have different patterns.

The prevailing winds in the winter period result in westerly transport of air masses across the U.S. On average, this results in on-shore flow over the western States, along the Texas Gulf Coast and the east coast of Florida. The polar jet stream is a prominent feature over the U.S. in the winter and as a result, the wind fields tend to be most dynamic in this

[EE] 2008 featured a waning La Nina phase of the ENSO as determined by the NOAA Climate Prediction Center. (http://www.cpc.ncep.noaa.gov/products/analysis_monitoring/ensostuff/ensoyears.shtml). Mean temperatures and precipitation patterns in 2008 were generally near long-term averages, with the exception of the Upper Midwest which was cooler and wetter than normal as determined by the NOAA National Climatic Data Center. (http://www.ncdc.noaa.gov/oa/climate/research/2008/cmb-prod-us-2008.html)

period. The wind fields around strong low pressure cyclones can advect air masses large distances (i.e., across the continent) in relatively short periods (i.e., less than a week).

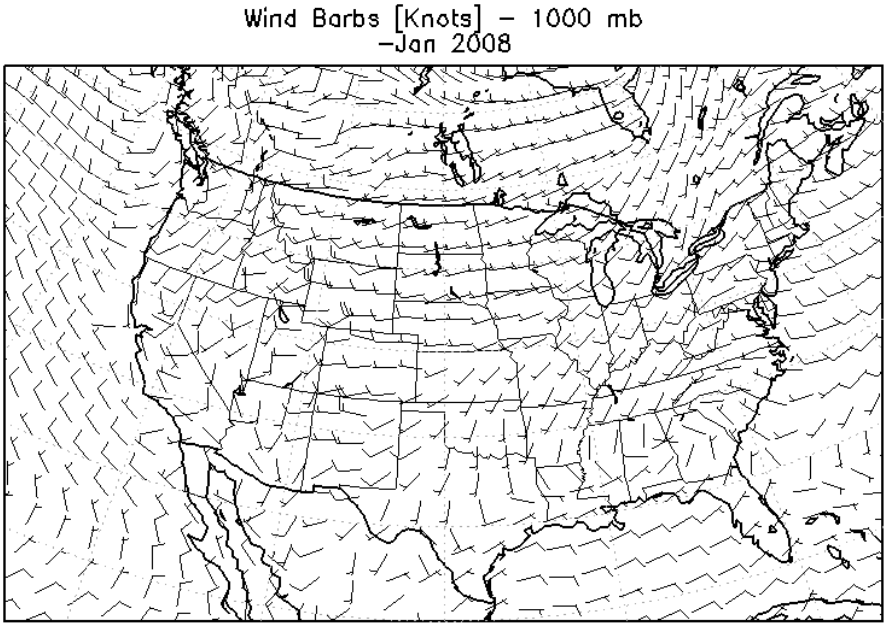

Figure 3A-1: Monthly Mean Winds in January 2008 Based on the NCEP Regional Reanalysis Dataset

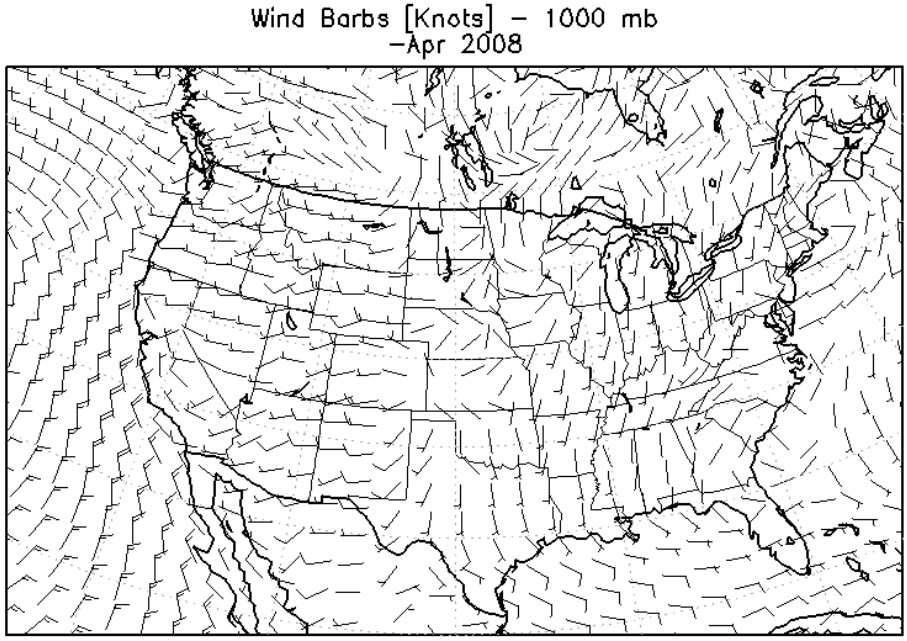

Figure 3A-2: Monthly Mean Winds in April 2008 Based on the NCEP Regional Reanalysis Dataset.

By the spring period, the mean wind flow still tends to be onshore over the Pacific Northwest, but it takes on a more parallel-to-the-coast alignment across California as a strong eastern Pacific anticyclone begins to set up. Along the Gulf Coast, southerly winds are common during this period. Strong low-level jet streams frequently originate over the

Midwestern U.S. during the spring resulting in the rapid northward advection of moist tropical air from the Gulf of Mexico to parts of the U.S. otherwise well removed from maritime influences. The mean wind fields are weak along the Atlantic Coast indicating near equal onshore/offshore winds. Although along the highly populated portions of the East Coast (New York, Philadelphia, Baltimore, Washington DC) there was a net tendency for transport off the ocean.

The eastern Pacific ridge is strong in the summertime and the prevailing winds tend to run along the West Coast. In immediate coastal environs it is common for diurnally-based wind patterns such as sea, land, bay, and lake breezes to govern how much onshore/offshore exchange takes place. The polar jet stream is typically located well north of the U.S./Canada border during the summer. Conditions tend to be more stagnant in this period than other times of the year. However, mean southerly winds over the Central U.S. expose large parts of the country to impacts from pollutants emitted or formed in the Gulf of Mexico. Mean winds around the Bermuda High that typically governs flows in the western Atlantic, generally results in offshore winds over the Eastern U.S. except in far north-eastern States like Connecticut, Massachusetts, and Maine where on average there is a considerable onshore wind component.

The fall season is a transition back to winter. Onshore winds begin to be more commonplace in Washington and Oregon. Subtropical trade winds result in low-level steering of air masses (and the occasional hurricane) into the Southeastern U.S. The predominant winds over the Northeastern U.S. are offshore as cold frontal passages from Canada become more frequent as the polar jet is displaced southward.

As noted earlier, there can be daily deviations within the prevailing seasonal winds. One tool that can be used to determine the origination of an air mass for a pollution event are Lagrangian trajectory models like HYSPLIT (Draxler and Hess, 1997) which calculates the path a plume of emissions would take given an input meteorological field. A set of three sample HYSPLIT 48-hour back trajectories are shown in Figure 3A-5 for a chosen day in the summer of 2008 with elevated levels of $PM_{2.5}$ over parts of the U.S. These figures are intended to provide a visual for what the HYSPLIT output products look like, more than to imply any causality between these particular trajectories and the resultant air quality on this day. The CMAQ air quality modeling, discussed above in Chapter 3.2.5, was used to isolate and estimate the impacts of shipping emissions on locations on land. These particular sample back-trajectories show a relatively stagnant atmosphere over Los Angeles with potential interactions with emissions from shipping sources just offshore. The back-trajectories over Birmingham and Philadelphia indicate that there is no direct maritime influence over the past two days for those locations. Of course, it is still possible that the longer-trajectories might indicate some small contribution to the overall background from sources over the water.

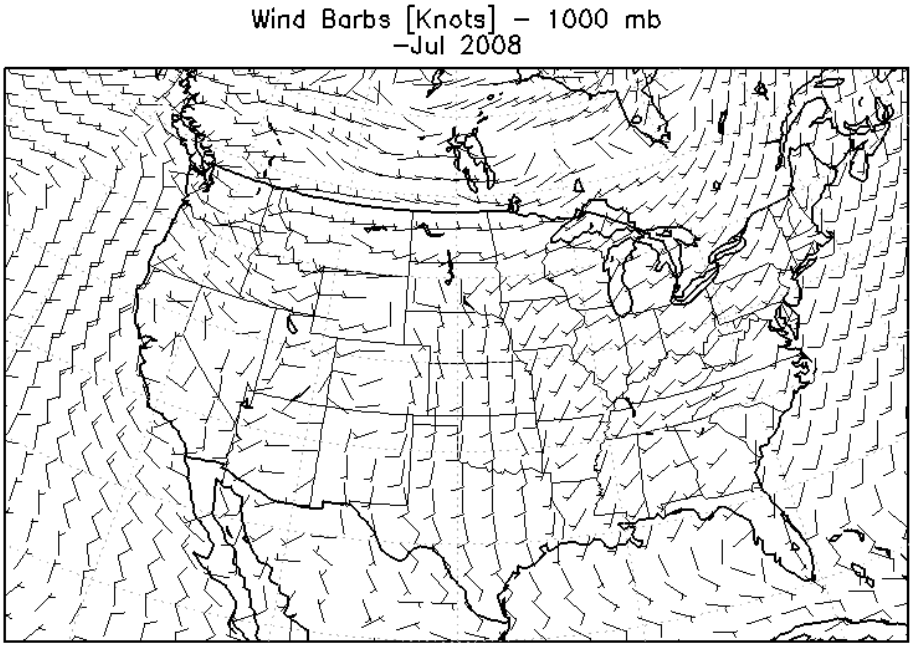

Figure 3A-3: Monthly Mean Winds in July 2008 Based on the NCEP Regional Reanalysis Dataset

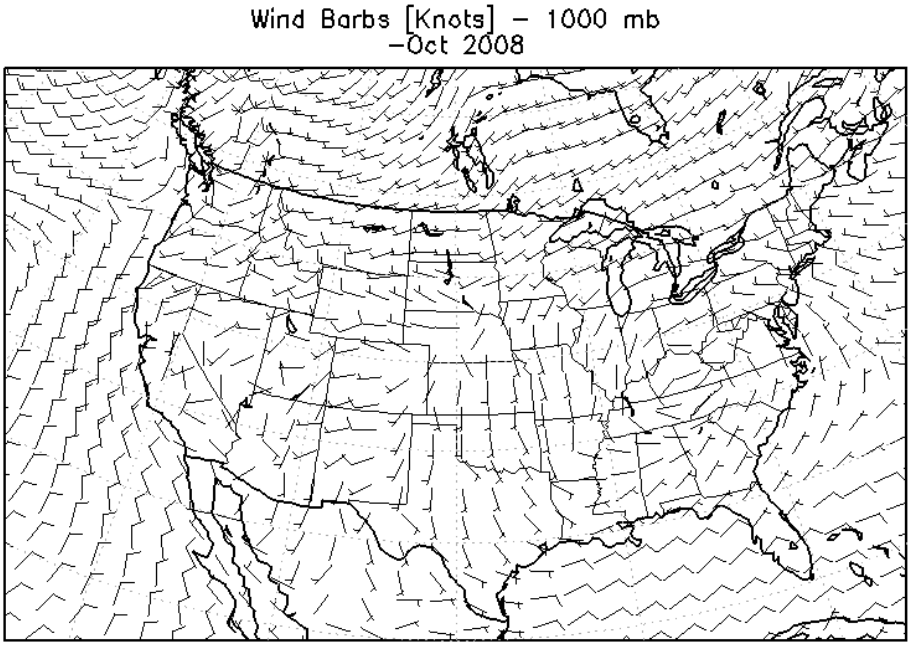

Figure 3A-4: Monthly Mean Winds in October 2008 Based on the NCEP Regional Reanalysis Dataset.

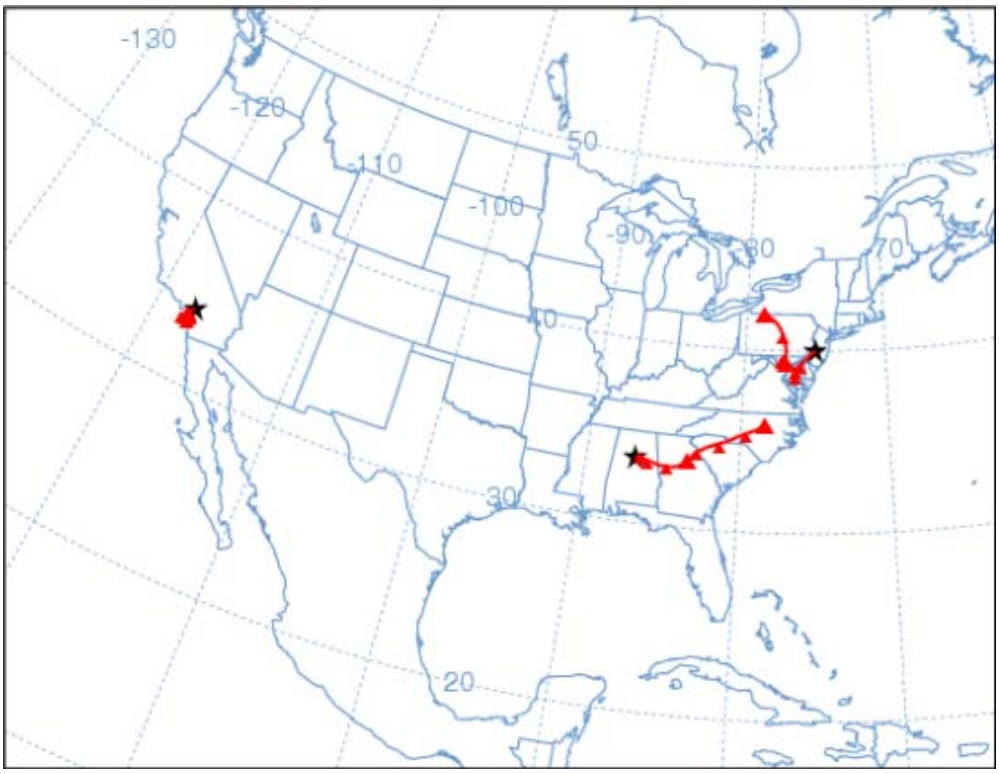

Figure 3A-5: 48-Hour Back-Trajectories from the HYSPLIT Trajectory Model. The red triangles represent how the air parcel that resided over the starred locations on 0000 GMT July 19, 2008 travelled over the preceding two days, in three hour increments.

Figure 3A-5 shows the compilation of daily (1800 GMT) 24-hour back trajectories over Los Angeles as derived from 12 years (1995-2006) of meteorological data provided by the Eta Data Assimilations System. For this location, if the mean transport direction (as determined from the starting point to the ending point of the trajectory) was from 150 to 300 degrees, then that day was flagged as potentially having a maritime influence. This analysis was completed for several major U.S. population centers near a coast. The results are shown in Table 3A-1. As can be seen, while the frequency of maritime influences can vary by location, it is not uncommon for locations all across the United States to be potentially affected by emissions that originate offshore.

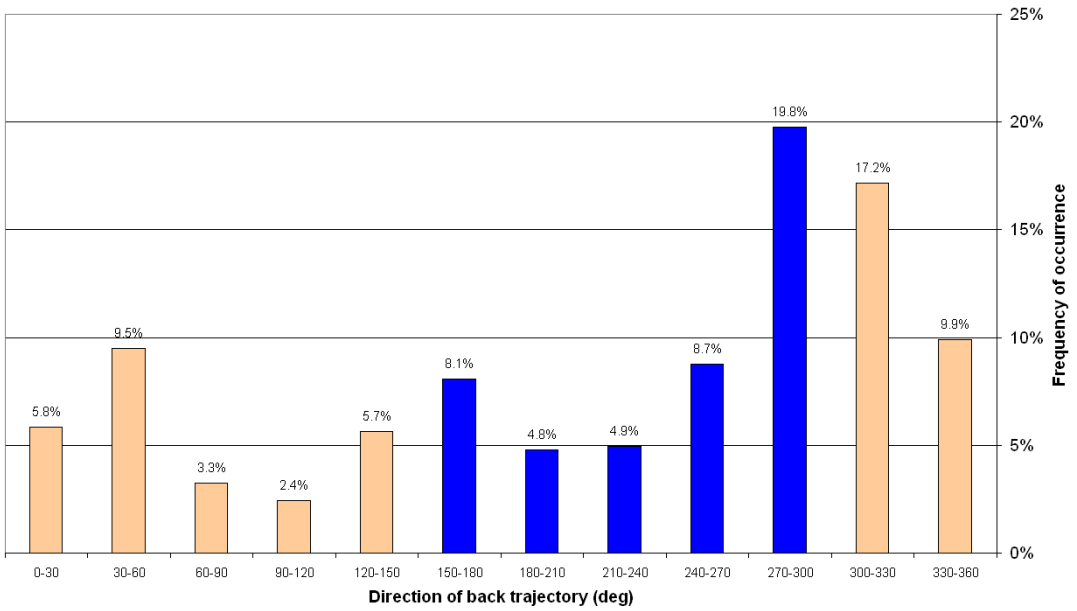

Figure 3A-6: 24-Hour Back Trajectory Directions in Los Angeles as Estimated by the HYSPLIT Model over the Period from 1995 to 2006

Table 3A-1: Summary of HYSPLIT back trajectories at highly-populated urban USA areas over a 12-year period showing the frequency at which the air mass likely emanated from a marine environment.

HIGHLY POPULATED USA COASTAL CITY	TRAJECTORY DIRECTIONS CONSIDERED TO BE INDICATIVE OF MARINE AIR (DEG)	FREQUENCY OF MARINE INTRUSION OVER THE PERIOD 1995-2006 (%)
San Francisco	180-330	45.7
Los Angeles	150-300	46.3
San Diego	180-330	67.2
Houston	90-210	58.9
New Orleans	90-240	48.7
Miami	30-180	65.8
New York City	30-180	19.0
Boston	30-120	12.5

In addition to the prevailing winds, the atmospheric stability can also conspire to result in land-based impacts from ship plumes. At certain locations and times of the year, the marine environment is characterized by a shallow temperature inversion (250-500m AGL) caused by the interaction between warmer subsiding air over cooler water (Winant et al., 1988). When ship emissions are injected into this shallow boundary layer, especially concentrated plumes can be maintained for long distances. This effect can be occasionally be seen in satellite pictures when clouds are formed by the exhaust from ships. When a persistent marine inversion exists, these clouds (and by extension the pollutant plumes from the ships) can be maintained for hundreds of kilometers and several days as shown below.

Figure 3A-7. MODIS Satellite Picture from May 11, 2005 Showing Clouds Formed from Ship Tracks. This public domain photo is from NASA's Earth Observatory at the website: http://earthobservatory.nasa.gov/IOTD/view.php?id=5488.

The MM5 meteorological modeling (Grell, et al., 1994) that was used to drive the air quality modeling simulations performed for this analysis captured this effect over the Eastern Pacific Ocean, the Northwest Atlantic Ocean, and the Great Lakes. Monthly average mixing heights over these regions were typically less than 300 m in the summer. This marine inversion prevents the ship plumes from being diluted vertically until they reach the coastal environs adjacent to the cool waters.

The last key meteorological element that is particularly relevant to any consideration of shipping emissions on human health and ecosystems is acid deposition. Deposition processes can occur in two modes: dry and wet. Wet deposition occurs when gases or particles are 'washed' out of the air by rain, snow, fog, or some other form of precipitation. The amount of precipitation over the water bodies surrounding North America can vary by location and season depending upon the synoptic meteorological patterns. However, orographical influences along the Pacific Northwest, and to a lesser extent over interior regions (e.g., Rocky Mountains, Appalachian Mountains) can lead to enhanced precipitation in those regions when the winds are from the ocean. Figure 3A-8 shows the monthly precipitation patterns over the U.S. for January 2008. When moist westerly winds are lifted up over the Cascade mountain range from Northern California through Washington State, large amounts of precipitation can occur on the windward side of the mountains. Additionally, in the summertime it is common for precipitation to be enhanced in coastal areas due to sea-breeze thunderstorms as well as general proximity to the moisture source.

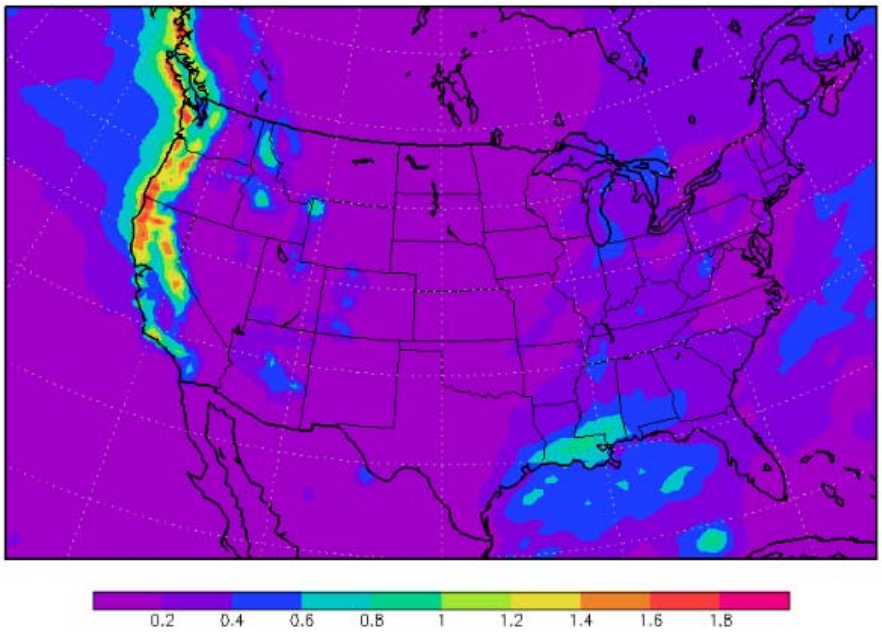

Figure 3A-8. Monthly Precipitation Accumulations in January 2008 from the NCEP Regional Reanalysis Dataset. Units are kg/m2.

The air quality modeling analyses and the meteorological discussion above focused on the 48-state contiguous portion of the United States, but the same meteorological conditions that result in potential impacts of ship emissions on air pollution over land in that region (e.g., prevailing winds, atmospheric stability, and precipitation patterns) can also result in potential impacts over Alaska and Hawaii. In fact, the oceanic influence is likely greater over the Hawaiian Islands and the coastal environs of Alaska (typically more populated than the interior portions of that State).

Because of its great expanse, the climatology of Alaska can differ widely depending upon latitude, altitude, and proximity to the ocean. Generally, the state's meteorology is classified in three zones: maritime, continental, and arctic. The weather in the maritime locations are strongly influenced by the relatively steady-state Pacific Ocean and as a results there are relatively small variations in prevailing winds, humidity levels and temperatures by season and location (Alaska Climate Research Center, 2009). Without the stabilizing influence of the ocean waters, the continental and arctic regions can experience large seasonal extremes in temperature, humidity, precipitation, and wind direction. The local meteorology in these two zones is driven by the topography of the surrounding areas, the altitude, and the fraction of sea ice in the Arctic Ocean.

The proximity of the maritime regions to the shipping lanes lead to the conclusion that populations in these areas would be most likely to be adversely impacted by air pollution originating from ships. While wind directions at measuring sites in Alaska can be strongly influenced by topography, the winds typically have an easterly component in populated locations like Anchorage, Juneau, Sitka, and Kenai (Western Regional Climate Center, 2009). Figure 3A-9 shows the average prevailing wind direction at 850 mb (approximately 1500 m above ground level) for the months of January and July, averaged over a recent 17 year period. The steering winds at this level indicate the potential for the transport of shipping

emissions in the North Pacific (shipping routes from Asia to North America). These winds are driven by common synoptic features that govern weather in this region, specifically the Aleutian low pressure cyclone in the winter and a northeastern Pacific anticyclone in the summer.

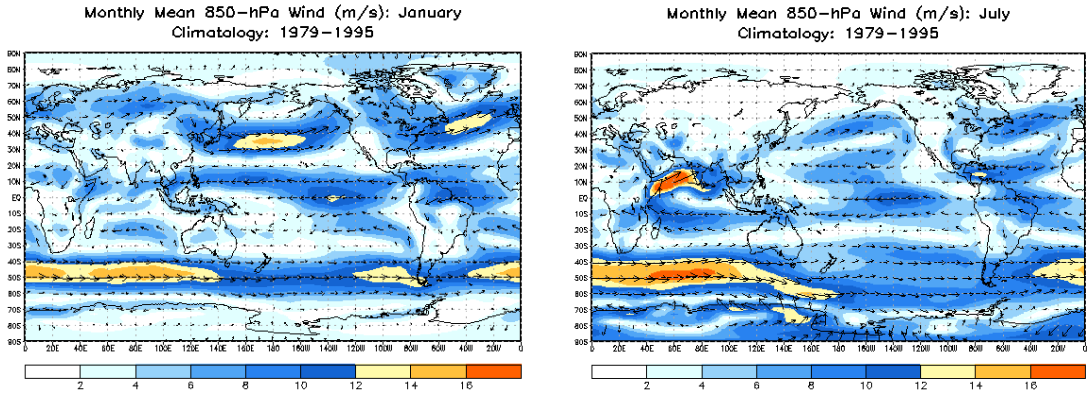

Figure 3A-9. Monthly Mean Winds at Approximately the 1,500 Meter Level in January (left) and July (right) Averaged over the Period from 1979 to 1995. Figures from NOAA Climate Prediction Center

Not surprisingly, Hawaiian meteorology is also subject to strong maritime influences. Kodama and Businger (1998) summarized the basic meteorology that occurs over this region. Global circulations such as the Hadley cell establish east-northeasterly trade winds as the predominant flow pattern in Hawaii, especially in the warm season. These trade winds can comprise 50-90 percent of the hourly wind directions over the region. Typically, the average height of the surface layer ranges from 1500-3000 m AGL in all seasons in Hawaii. Any emissions input to this layer will remain in this layer unless ventilated by convection or removed by deposition. Ultimately, as there are shipping lanes on all sides of the main Hawaiian Islands; regardless of which way the wind blows, there is a high potential for ship emissions to affect air pollution over land.

In conclusion, there is ample evidence that the meteorological conditions in the proposed area of application have the potential to put human populations and environmental areas at risk of adverse environmental impacts from ship emissions. This conclusion is confirmed by the air quality modeling analyses performed for this assessment.

APPENDIX 3B

Table 3B-1. Percent reduction in Nitrogen (N) and Sulfur (S) deposition averaged over a 2-digit HUC sub region for two modeling scenarios. The range of reductions for individual HUCs within the sub region is shown in parentheses.

HUC SUB REGION		ZERO C3 EMISSIONS	ECA
New England (1)	average reduction (range) in N deposition	4.9% (2.6 to 11.0%)	1.3% (0.7 to 3.5%)
	average reduction (range) in S deposition	6.3% (3.0 to 16.3%)	5.3% (1.8 to 15.0%)
Mid Atlantic (2)	average reduction (range) in N deposition	3.1% (1.1 to 7.4%)	0.8% (0.1 to 1.9%)
	average reduction (range) in S deposition	6.6% (1.2 to 14%)	6.0% (1.0 to 13.0%)
South Atlantic - Gulf (3)	average reduction (range) in N deposition	5.9% (1.8 to 11.4%)	1.1% (0.3 to 2.8%)
	average reduction (range) in S deposition	8.7% (3.1 to 10.3%)	6.1% (2.0 to 7.1%)
Great Lakes (4)	average reduction (range) in N deposition	0.9% (0.4 to 1.7%)	0.2% (0.1 to 0.5%)
	average reduction (range) in S deposition	1.2% (0.6 to 2.9%)	1.0% (0.5 to 2.7%)
Ohio (5)	average reduction (range) in N deposition	1.5% (0.6 to 2.5%)	0.4% (0.1 to 0.7%)
	average reduction (range) in S deposition	1.4% (0.8 to 3.3%)	1.0% (0.6 to 2.2%)
Tennessee (6)	average reduction (range) in N deposition	2.5% (0.6 to 3.8%)	0.6% (0.1 to 1.0%)
	average reduction (range) in S deposition	2.8% (0.8 to 5.0%)	1.9% (0.6 to 3.5%)
Upper Mississippi (7)	average reduction (range) in N deposition	0.5% (0.2 to 1.4%)	0.1% (0.1 to 0.4%)
	average reduction (range) in S deposition	1.1% (0.4 to 2.2%)	0.7% (0.3 to 1.3%)
Lower Mississippi (8)	average reduction (range) in N deposition	5.1% (2.6 to 11.5%)	1.2% (0.5 to 2.8%)
	average reduction (range) in S deposition	7.8% (4.5 to 15.6%)	5.8% (3.2 to 11.3%)
Souris-Red-Rainy (9)	average reduction (range) in N deposition	0.3% (0.2 to 17.2%)	0.1% (0.1 to 4.8%)
	average reduction (range) in S deposition	0.9% (0.3 to 33.3%)	0.6% (0.2 to 28.5%)
Missouri (10)	average reduction (range) in N deposition	0.6% (0.4 to 1.8%)	0.2% (0.1 to 0.5%)
	average reduction (range) in S deposition	1.8% (1.3 to 3.7%)	1.1% (0.7 to 2.2%)

HUC SUB REGION		ZERO C3 EMISSIONS	ECA
Arkansas-White-Red (11)	average reduction (range) in N deposition	1.5% (0.6 to 6.8%)	0.3% (0.1 to 1.7%)
	average reduction (range) in S deposition	3.6% (1.6 to 7.6%)	2.2% (0.8 to 5.4%)
Texas-Gulf (12)	average reduction (range) in N deposition	3.3% (1.7 to 7.7%)	0.5% (0.0 to 1.4%)
	average reduction (range) in S deposition	7.0% (2.3 to 11.7%)	4.9% (1.3 to 8.4%)
Rio Grande (13)	average reduction (range) in N deposition	2.0% (0.7 to 2.9%)	0.4% (0.2 to 0.5%)
	average reduction (range) in S deposition	3.2% (1.5 to 4.4%)	1.7% (0.8 to 2.4%)
Upper Colorado (14)	average reduction (range) in N deposition	1.6% (1.2 to 3.1%)	0.6% (0.5 to 1.2%)
	average reduction (range) in S deposition	2.8% (1.0 to 7.1%)	2.2% (0.8 to 5.6%)
Lower Colorado (15)	average reduction (range) in N deposition	3.3% (1.7 to 5.5%)	0.9% (0.4 to 1.5%)
	average reduction (range) in S deposition	5.2% (3.2 to 10.1%)	3.3% (1.6 to 7.4%)
Great Basin (16)	average reduction (range) in N deposition	2.0% (1.2 to 3.0%)	0.8% (0.5 to 1.5%)
	average reduction (range) in S deposition	4.4% (2.1 to 7.1%)	3.7% (1.7 to 6.1%)
Pacific Northwest (17)	average reduction (range) in N deposition	4.9% (2.2 to 33.5%)	1.0% (0.1 to 6.1%)
	average reduction (range) in S deposition	14.5% (5.1 to 56.4%)	11.1% (4 to 37.5%)
California (18)	average reduction (range) in N deposition	8.4% (2.5 to 40.4%)	2.3% (0.7 to 13.4%)
	average reduction (range) in S deposition	21.3% (4.6 to 81.6%)	19.4% (3.8 to 78.1%)

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