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

Technical Support Document

United States
Environmental Protection
Agency

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Assessment and Standards Division
Office of Transportation and Air Quality
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1 Executive Summary

Introduction

On March 27, 2009, the United States and Canada submitted a joint proposal (MEPC 59/6/5) to the International Maritime Organization to designate an Emission Control Area (ECA) for specific portions of U.S. and Canadian coastal waters. This action would control emissions of nitrogen oxides (NO_X), sulfur oxides (SO_X), and particulate matter (PM) from ships. Designation of the proposed ECA is necessary to protect public health and the environment in the United States and Canada by reducing exposure to harmful levels of air pollution resulting from these emissions. The burden on international shipping is small compared to the improvements in air quality, reductions in premature mortality and other benefits resulting from designation of the proposed ECA.

This Technical Support Document provides a comprehensive presentation of the many in-depth technical analyses performed by the U.S. Government, in developing the ECA proposal.

Emission Inventory

Chapter 2 describes how U.S. emission inventories were developed to describe air emissions from ships operating in waters within the proposed ECA. These inventories provide the foundation upon which all the subsequent analyses were built, and address Criterion 6 of Section 3, Appendix III to MARPOL Annex VI. Beyond the level of detail provided in MEPC 59/6/5, Chapter 2 explains how the inputs were developed and what assumptions were made in assessing what the emissions are from ships currently (2002 base year), what the emissions would look like in 2020 without the proposed ECA, and what reductions can be expected from the proposed ECA.

Chapter 2 describes the “bottom-up” methodology that was used, based on the latest state of the art models and inputs. This chapter describes which port-related emissions were included and why, and how emissions were obtained for ships while underway in U.S. waters. This chapter explains in great detail each parameter that went into the modeling and analyses, including which ships are included, which fuels are used by those ships, which other (non-ECA) emission controls are in place for each scenario, and what growth rates are expected, incorporating forecasts of the demand for marine transportation services in 2020.

Impacts of Emissions on Air Quality, Human Health and the Environment

Chapter 3 describes in great detail most of the analyses conducted in support of Criteria 2, 3, 4 and 5 of Section 3, Appendix III to MARPOL Annex VI. For organizational reasons, the analyses conducted to assess the impacts of ships’ emissions on human health are presented in Chapter 4, summarized below. Chapter 3 contains several sub-sections, outlined here for ease of reference.

Impacts of Pollutants on Human Health

Section 3.1 describes the human health impacts of the pollutants proposed for control in the U.S./Canada ECA. The proposed ECA would not only reduce direct emissions of NO_X, SO_X and PM, but also secondarily formed ambient PM and ground-level ozone. Section 3.1.1 describes the nature of these pollutants, formation processes, and relationship to ship emissions. Section 3.1.2 presents the health effects associated with exposure to NO_X, SO_X, PM and ground-level ozone, summarizing the key scientific literature.

Impacts of Ships' Emissions on Air Quality and Benefits of ECA to Air Quality

Section 3.2 describes the effects of NO_X, SO_X and PM emissions on ambient air quality under the same scenarios for which emission inventories were developed, presented in terms of ground-level ozone and PM. This section also describes the multi-pollutant modeling platform that was used to assess the impacts of reduced marine emissions from the application of the proposed ECA. Appendix A to Chapter 3 describes the relevant meteorological conditions that contribute to at-sea emissions being transported to populated areas and contributing to harmful human health and ecological impacts, and which formed inputs to the modeling platform.

Impacts of Ships' Emissions on Ecosystems and Benefits of ECA to Ecosystems

Section 3.3 describes the impacts of emissions from ships on terrestrial and aquatic ecosystems such as visibility, ozone uptake, eutrophication, acidification, loss of forest biomass, and overall forest health. Using the same scenarios as for the other analyses, improvements in environmental conditions for many types of ecosystems were evaluated. Unlike the analyses for human health, there are a larger number of pollutants of concern to ecosystems. Thus, deposition of many chemical forms of NO_X, SO_X and PM are discussed in this section, as well as the biogeochemical cycles of interrelated pollutants such as mercury.

Impacts of Ships' Emissions on Human Health and Benefits of ECA to Human Health

Chapter 4 presents quantified U.S.-related health impacts for PM and ozone associated with emissions from ships, both in terms of the expected contribution of overall ship emissions to adverse health impacts on land and the reductions in adverse health impacts that can be expected to occur from the adoption of the proposed ECA.

The health impacts modeling presented in Chapter 4 is based on peer-reviewed studies of air quality and health and welfare effects associated with improvements in air quality. This chapter also describes the computer program used to estimate health benefits by integrating a number of modeling elements (e.g., interpolation functions, population projections, health impact functions, valuation functions, analysis and pooling methods) to translate modeled air concentration estimates into health effect incidence estimates.

Cost Analyses

Chapter 5 describes our estimates of the costs associated with the reduction of SO_X, NO_X, and PM emissions from ships, not only to the shipping industry but also to marine fuel suppliers and companies who rely on the shipping industry. This chapter provides additional detail regarding the analyses conducted in support of Criteria 7 and 8 of Section 3, Appendix III to MARPOL Annex VI. This chapter describes the analyses used to evaluate the cost impact of Tier III NO_X requirements combined with low sulfur fuel use on vessels operating within the proposed ECA, including estimates of low sulfur fuel production costs, vessel hardware costs, and operating costs. This chapter also presents cost per ton estimates for ECA-based NO_X and fuel sulfur standards and compares these with the costs of established land-based control programs.

Economic Impact Analysis

Chapter 6 examines the economic impacts of the projected ECA costs on shipping engaged in international trade. This chapter provides additional detail in support of Criterion 8 of Section 3, Appendix III to MARPOL Annex VI. This chapter describes the econometric methodology that was used in estimating two aspects of the economic impacts: social costs and how they are shared across stakeholders, and market impacts for the new engine and new vessel markets.

2 Emission Inventory

2.1 Introduction

Ships (i.e., ocean-going vessels) are significant contributors to the total United States (U.S.) mobile source emission inventory. The U.S. ship inventory reported here focuses on Category 3 (C3) vessels, which use C3 engines for propulsion. C3 engines are defined as having displacement above 30 liters per cylinder (L/cyl). The resulting inventory includes emissions from both propulsion and auxiliary engines used on these vessels, as well as those on gas and steam turbine vessels.

Most of the vessels operating in U.S. ports that have propulsion engines less than 30 liters per cylinder are domestic and are already subject to strict national standards affecting NO_X, PM, and fuel sulfur content. As such, the inventory does not include any ships, foreign or domestic, powered by Category 1 or Category 2 (i.e., <30 L/cyl) engines. In addition, as discussed in Sections 2.3.2.3.9 and 2.3.3.3, this inventory is primarily based on activity data for ships that carry foreign cargo. Category 3 vessels carrying domestic cargo that operate only between U.S. ports are only partially accounted for in this inventory.[1] Emissions due to military vessels are also excluded.

The regional and national inventories for C3 vessels presented in this chapter are sums of independently constructed port and interport emissions inventories. Port inventories were developed for 89 deep water and 28 Great Lake ports in the U.S.[2] While there are more than 117 ports in the U.S., these are the top U.S. ports in terms of cargo tonnage. Port-specific emissions were calculated with a “bottom-up” approach, using data for vessel calls, emission factors, and activity for each port. Interport emissions and emissions for the remaining ports were obtained using the Waterway Network Ship Traffic, Energy and Environment Model (STEEM).[3,4] STEEM also uses a “bottom-up” approach, estimating emissions from C3 vessels using historical North American shipping activity, ship characteristics, and activity-based emission factors. STEEM was used to quantify and geographically (i.e., spatially) represent interport vessel traffic and emissions for vessels traveling generally within 200 nautical miles (nm) of the U.S.

The detailed port inventories were spatially merged into the STEEM gridded inventory to create a comprehensive inventory for Category 3 vessels. For the 117 ports, this involved removing the near-port portion of the STEEM inventory and replacing it with the detailed port inventories. For the remaining U.S. ports for which detailed port inventories are not available, the near-port portion of the STEEM inventory was simply retained. This was done for a base year of 2002. Inventories for 2020 were then projected using regional growth rates[5,6] and adjustment factors to account for the International Maritime Organization (IMO) Tier I and Tier II NO_X standards and NO_X retrofit program.[2] Inventories incorporating additional Tier III NO_X and fuel sulfur controls within the proposed Emission Control Area (ECA) were also developed for 2020.

The inventory estimates reported in this chapter include emissions out to 200 nm from the U.S. coastline, including Alaska and Hawaii, but not extending into the Exclusive Economic Zone (EEZ) of neighboring countries. Inventories are presented for the following pollutants: oxides of nitrogen (NO_X), particulate matter ($PM_{2.5}$ and PM_{10}), sulfur dioxide (SO_2), hydrocarbons (HC), carbon monoxide (CO), and carbon dioxide (CO_2). The PM inventories include directly emitted PM only, although secondary sulfates and nitrates are taken into account in the air quality modeling.

2.2 Modeling Domain and Geographic Regions

The inventories described in this chapter reflect ship operations that occur within the area that extends 200 nautical miles (nm) from the official U.S. baseline but exclude operations in Exclusive Economic Zones of other countries. The official U.S. baseline is recognized as the low-water line along the coast as marked on the official U.S. nautical charts in accordance with the articles of the Law of the Sea. The boundary was mapped using geographic information system (GIS) shapefiles obtained from the National Oceanic and Atmospheric Administration, Office of Coast Survey.[7] The accuracy of the NOAA shapefiles was verified with images obtained from the U.S. Geological Survey. The confirmed NOAA shapefiles were then combined with a shapefile of the U.S. international border from the National Atlas.[8]

The resulting region was further subdivided for this analysis to create regions that were compatible with the geographic scope of the regional growth rates, which are used to project emission inventories for the years 2020, as described later in this document.

- The Pacific Coast region was split into separate North Pacific and South Pacific regions along a horizontal line originating from the Washington/Oregon border (Latitude 46° 15' North).
- The East Coast and Gulf of Mexico regions were divided along a vertical line roughly drawn through Key Largo (Longitude 80° 26' West).
- The Alaska region was divided into separate Alaska Southeast and Alaska West regions along a straight line intersecting the cities of Naknek and Kodiak. The Alaska Southeast region includes most of the State's population, and the Alaska West region includes the emissions from ships on a great circle route along the Aleutian Islands between Asia and the U.S. West Coast.
- For the Great Lakes domain, shapefiles were created containing all the ports and inland waterways in the near port inventory and extending out into the lakes to the international border with Canada. The modeling domain spanned from Lake Superior on the west to the point eastward in the State of New York where the St. Lawrence River parts from U.S. soil.
- The Hawaiian domain was subdivided so that a distance of 200 nm beyond the southeastern islands of Hawai'i, Maui, O'ahu, Moloka'i, Ni'ihau, Kaua'i, Lanai, and Kahoolawe was contained in Hawaii East. The remainder of the Hawaiian Region was then designated Hawaii West.

This methodology resulted in nine separate regional modeling domains that are identified below and shown in Figure 2-1. U.S. territories are not included in this analysis.

- South Pacific (SP)
- North Pacific (NP)
- East Coast (EC)
- Gulf Coast (GC)
- Alaska Southeast (AE)
- Alaska West (AW)
- Hawaii East (HE)

- Hawaii West (HW)
- Great Lakes (GL)

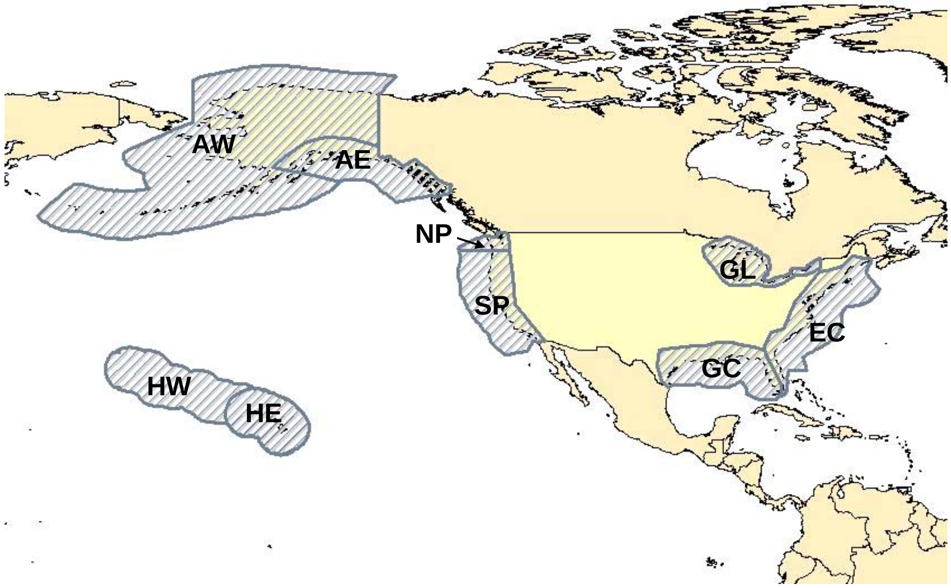


Figure 2-1 Regional Modeling Domains

2.3 Development of 2002 Inventories

This section describes the methodology and inputs, and presents the resulting inventories for the 2002 baseline calendar year. The first section describes the general methodology. The second section describes the methodology, inputs, and results for near port emissions. The third section describes the methodology and inputs for emissions when operating away from port (also referred to as "interport" emissions). The fourth section describes the method for merging the interport and near port portions of the inventory. Resulting total emissions for the U.S., as well as for nine geographic regions within the U.S., are then presented.

2.3.1 Outline of Methodology

The total inventory was created by summing emissions estimates for ships while at port (near port inventories) and while underway (interport inventories). Near port inventories for calendar year 2002 were developed for 117 U.S. commercial ports that engage in foreign trade. Based on an analysis of U.S. Government data, these 117 commercial ports encompass nearly all U.S. C3 vessel calls.[9]

The outer boundaries of the ports are defined as 25 nm from the terminus of the reduced speed zone for deep water ports and 7 nm from the terminus of the reduced speed zone for Great Lake ports. Port emissions are calculated for different modes of operation and then summed. Emissions for each mode are calculated using port-specific information for vessel calls, vessel characteristics, and activity, as well as other inputs that vary instead by vessel or engine type (e.g., emission factors).

The interport inventory is estimated using the Waterway Network Ship Traffic, Energy, and Environmental Model (STEEM).[3,4] The model geographically characterizes emissions from ships traveling along shipping lanes to and from individual ports, in addition to the emissions from vessels transiting near the ports. The shipping lanes were identified from actual ship positioning reports. The model then uses detailed information about ship destinations, ship attributes (e.g., vessel speed and engine horsepower), and emission factors to produce spatially allocated (i.e., gridded) emission estimates for ships engaged in foreign commerce.

The 117 near port inventories are an improvement upon STEEM's near port results in several ways. First, the precision associated with STEEM's use of ship positioning data may be less accurate in some locations, especially as the lanes approach shorelines where ships would need to follow more prescribed paths. Second, the STEEM model includes a maneuvering operational mode (i.e., reduced speed) that is generally assumed to occur for the first and last 20 kilometers of each trip when a ship is leaving or entering a port. In reality, the distance when a ship is traveling at reduced speeds varies by port. Also, the distance a ship traverses at reduced speeds often consists of two operational modes: a reduced speed zone (RSZ) as a ship enters or leaves the port area and actual maneuvering at a very low speed near the dock. Third, the STEEM model assumes that the maneuvering distance occurs at an engine load of 20 percent, which represents a vessel speed of approximately 60 percent of cruise speed. This is considerably faster than ships would maneuver near the docks. The single maneuvering speed assumed by STEEM also does not reflect the fact that the reduced speed zone, and therefore emissions, may vary by port. Fourth, and finally, the STEEM model does not include the emissions from auxiliary engines during hotelling operations at the port. The near-port inventories correct these issues.

The regional emission inventories produced by the current STEEM interport model are most accurate for vessels while cruising in ocean or Great Lakes shipping lanes; the near port inventories use more detailed local port information and are significantly more accurate near the ports. Therefore, the inventories in this analysis are derived by merging together: 1) the near port inventories, which extend 25 nautical miles and 7 nautical miles from the terminus of the RSZ for deep water ports and Great Lake ports, respectively, and 2) the remaining interport portion of the STEEM inventory, which extends from the endpoint of the near port inventories to the 200 nautical mile boundary or international border with Canada, as appropriate. Near some ports, a portion of the underlying STEEM emissions were retained if it was determined that the STEEM emissions included ships traversing the area near a port, but not actually entering or exiting the port.

2.3.2 Near Port Emissions

Near port inventories for calendar year 2002 were developed for ocean-going vessels at 89 deep water and 28 Great Lake ports in the U.S. The inventories include emissions from both propulsion and auxiliary engines on C3 vessels.

This section first describes the selection of the ports for analysis and then provides the methodology used to develop the near port inventories. This is followed by a description of the key inputs. Total emissions by port and pollutant for 2002 are then presented.

2.3.2.1 Selection of Individual Ports to be Analyzed

All 150 deep water and Great Lake ports in the Principal Ports of the United States dataset[10] were used as a starting point. Thirty ports which had no foreign traffic were eliminated because the dataset used to obtain port calls and other ship characteristics has no information about domestic traffic. Several California ports were also used because the California Air Resources Board (ARB) provided the necessary data and estimates for those ports. The final list of 117 deep water and Great Lake ports, along with their coordinates, is given in Appendix 2A.

2.3.2.2 Port Methodology

Near port emissions for each port are calculated for four modes of operation: 1) hotelling, 2) maneuvering, 3) reduced speed zone (RSZ), and 4) cruise. Hotelling, or dwelling, occurs while the vessel is docked or anchored near a dock, and only the auxiliary engine(s) are being used to provide power to meet the ship's energy needs. Maneuvering occurs within a very short distance of the docks. The RSZ varies from port to port, though generally the RSZ would begin and end when the pilots board or disembark, and typically occurs when the near port shipping lanes reach unconstrained ocean shipping lanes. The cruise mode emissions in the near ports analysis extend 25 nautical miles beyond the end of the RSZ lanes for deep water ports and 7 nautical miles for Great Lake ports.

Emissions are calculated separately for propulsion and auxiliary engines. The basic equation used is as follows:

Equation 2-1

$$Emissions_{\mathrm{mode}[eng]} = (calls) \times (P_{[eng]}) \times (hrs/call_{\mathrm{mode}}) \times (LF_{\mathrm{mode}[eng]}) \times (EF_{[eng]}) \times (Adj) \times (10^{-6}\ tonnes/g)$$

Where:

- $Emissions_{\mathrm{mode\ [eng]}}$ = Metric tonnes emitted by mode and engine type
- Calls = Round-trip visits (i.e., one entrance and one clearance is considered a call)
- $P_{[\mathrm{eng}]}$ = Total engine power by engine type, in kilowatts
- $hrs/call_{\mathrm{mode}}$ = Hours per call by mode
- $LF_{\mathrm{mode\ [eng]}}$ = Load factor by mode and engine type (unitless)
- $EF_{[\mathrm{eng}]}$ = Emission factor by engine type for the pollutant of interest, in g/kW-hr (these vary as a function of engine type and fuel used, rather than activity mode)
- Adj = Low load adjustment factor, unitless (used when the load factor is below 0.20)
- 10^{-6} = Conversion factor from grams to metric tonnes

Main engine load factors are calculated directly from the propeller curve based upon the cube of actual speed divided by maximum speed (at 100% maximum continuous rating [MCR]). In addition, cruise mode activity is based on cruise distance and speed inputs. Appendix 2B provides the specific equations used to calculate propulsion and auxiliary emissions for each activity mode.

2.3.2.3 Inputs for Port Emission Calculations

The following inputs are required to calculate emissions for the four modes of operation (cruise, RSZ, maneuvering, and hotelling):

- Number of calls
- Main engine power
- Cruise (vessel service) speed
- Cruise distance
- RSZ distance for each port
- RSZ speed for each port
- Auxiliary engine power
- Auxiliary load factors
- Main and auxiliary emission factors
- Low load adjustment factors for main engines
- Maneuvering time-in-mode (hours/call)
- Hotelling time-in-mode (hours/call)

Note that load factors for main engines are not listed explicitly, since they are calculated as a function of mode and/or cruise speed. This section describes the inputs in more detail, as well as the sources for each input.

2.3.2.3.1 Calls and Ship Characteristics (Propulsion Engine Power and Cruise Speed)

For this analysis, U.S. Army Corps of Engineers (USACE) entrance and clearance data for 2002,[11] together with Lloyd's data for ship characteristics,[12] were used to calculate average ship characteristics and calls by ship type for each port. Information for number of calls, propulsion engine power, and cruise speed were obtained from these data.

2.3.2.3.1.1 Bins by Ship Type, Engine Type, and DWT Range

The records from the USACE entrances and clearances data base were matched with Lloyd's data on ship characteristics for each port. Calls by vessels that have either Category 1 or 2 propulsion engines were eliminated from the data set. The data was then binned by ship type, engine type and dead weight tonnage (DWT) range. The number of entrances and clearances in each bin are counted, summed together and divided by two to determine the number of calls (i.e., one entrance and one clearance was considered a call). For Great Lake ports, there is a larger frequency of ships either entering the port loaded and leaving unloaded (light) or entering the port light and leaving loaded. In these cases, there would only be one record (the loaded trip into or out of the port) that would be present in the data. For Great Lake ports, clearances were matched with entrances by ship name. If there was not a reasonable match, the orphan entrance or clearance was treated as a call.

Propulsion power and vessel cruise speed are also averaged for each bin. Auxiliary engine power was computed from the average propulsion power using the auxiliary power to propulsion power ratios discussed in section 2.3.2.3.4.

2.3.2.3.1.2 Removal of Category 1 and 2 Ships

Since these inventories were intended to cover Category 3 propulsion engine ships only, the ships with Category 1 and 2 propulsion engines were eliminated. This was accomplished by matching all ship calls with information from Lloyd's Data, which is produced by Lloyd's Register-Fairplay Ltd.[12] Over 99.9 percent of the calls in the entrances and clearances data were directly matched with Lloyd's data. The remaining 0.1 percent was estimated based upon ships of similar type and size.

Engine category was determined from engine make and model. Engine bore and stroke were found in the Marine Engine 2005 Guide[13] and displacement per cylinder was calculated. Ships with Category 1 or 2 propulsion engines were eliminated from the data.

Many passenger ships and tankers have either diesel-electric or gas turbine-electric engines that are used for both propulsion and auxiliary purposes. Both were included in the current inventory.

2.3.2.3.2 Cruise Distance

Cruise mode emissions are calculated assuming a 25 nautical mile distance into and out of the port for deep water ports and 7 nautical miles into and out of the port for Great Lake ports outside of the reduced speed and maneuvering zones.

2.3.2.3.3 RSZ Distances and Speeds by Port

Reduced speed zone (RSZ) distance and speed were individually determined for each port. For deep water ports, the RSZ distances were developed from shipping lane information contained in the USACE National Waterway Network.[14] The database defines waterways as links or line segments that, for the purposes of this study, represent actual shipping lanes (i.e., channels, intracoastal waterways, sea lanes, and rivers). The sea-side endpoint for the RSZ was selected as the point along the line segment that was judged to be far enough into the ocean where ship movements were unconstrained by the coastline or other vessel traffic. The resulting RSZ distance was then measured for each deep water port. The final RSZ distances and endpoints for each port are listed in Appendix 2C. The RSZ for each Great Lake port was fixed at three nautical miles.

2.3.2.3.4 Auxiliary Engine Power and Load Factors

Since hotelling emissions are a large part of port inventories, it is important to distinguish propulsion engine emissions from auxiliary engine emissions. In the methodology used in this analysis, auxiliary engine maximum continuous rating power and load factors were calculated separately from propulsion engines and different emission factors (EFs) applied. All auxiliary engines were treated as Category 2 medium-speed diesel (MSD) engines for purposes of this analysis.

Auxiliary engine power is not contained in the USACE database and is only sparsely populated in the Lloyd's database; as a result, it must be estimated. The approach taken was to derive ratios of average auxiliary engine power to propulsion power based on survey data. The California Air Resources Board (ARB) conducted an Oceangoing Ship Survey of 327 ships in January 2005 that was principally used for this analysis.[15] Average auxiliary engine power to propulsion power ratios were estimated by ship type and are presented in Table 2-1. These ratios by ship type were applied to the propulsion power data to derive auxiliary power for the ship types at each port.

Table 2-1 Auxiliary Engine Power Ratios (ARB Survey, except as noted)

Ship Type	Average Propulsion Engine (kW)	Average Auxiliary Engines Number	Power Each (kW)	Total Power (kW)	Engine Speed	Auxiliary to Propulsion Ratio
Auto Carrier	10,700	2.9	983	2,850	Medium	0.266
Bulk Carrier	8,000	2.9	612	1,776	Medium	0.222
Container Ship	30,900	3.6	1,889	6,800	Medium	0.220
Passenger Ship[a]	39,600	4.7	2,340	11,000	Medium	0.278
General Cargo	9,300	2.9	612	1,776	Medium	0.191
Miscellaneous[b]	6,250	2.9	580	1,680	Medium	0.269
RORO	11,000	2.9	983	2,850	Medium	0.259
Reefer	9,600	4.0	975	3,900	Medium	0.406
Tanker	9,400	2.7	735	1,985	Medium	0.211

[a] Many passenger ships typically use a different engine configuration known as diesel-electric. These vessels use large generator sets for both propulsion and ship-board electricity. The figures for passenger ships above are estimates taken from the Starcrest Vessel Boarding Program.

[b] Miscellaneous ship types were not provided in the ARB methodology, so values from the Starcrest Vessel Boarding Program were used.

Auxiliary engine to propulsion engine power ratios vary by ship type and operating mode roughly from 0.19 to 0.40. Auxiliary load, shown in Table 2-2, is used together with the total auxiliary engine power to calculate auxiliary engine emissions. Starcrest's Vessel Boarding Program[16] showed that auxiliary engines are on all of the time, except when using shoreside power during hotelling.

Table 2-2 Auxiliary Engine Load Factor Assumptions

Ship-Type	Cruise	RSZ	Maneuver	Hotel
Auto Carrier	0.13	0.30	0.67	0.24
Bulk Carrier	0.17	0.27	0.45	0.22
Container Ship	0.13	0.25	0.50	0.17
Passenger Ship	0.80	0.80	0.80	0.64
General Cargo	0.17	0.27	0.45	0.22
Miscellaneous	0.17	0.27	0.45	0.22
RORO	0.15	0.30	0.45	0.30
Reefer	0.20	0.34	0.67	0.34
Tanker	0.13	0.27	0.45	0.67

2.3.2.3.5 Fuel Types and Fuel Sulfur Levels

There are primarily three types of fuel used by marine engines: residual marine (RM), marine diesel oil (MDO), and marine gas oil (MGO), with varying levels of fuel sulfur.[5] MDO and MGO are generally described as distillate fuels. For this analysis, RM and MDO fuels are assumed to be used. Since PM and SO_2 emission factors are dependent on the fuel sulfur level, calculation of port inventories requires information about the fuel sulfur levels associated with each fuel type, as well as which fuel types are used by propulsion and auxiliary engines.

Based on an ARB survey,[15] average fuel sulfur level for residual marine was set to 2.5 percent for the west coast and 2.7 percent for the rest of the country. A sulfur content of 1.5 percent was used for MDO.[17] While a more realistic value for MDO used in the U.S. appears to be 0.4 percent, given the small proportion of distillate fuel used by ships relative to RM, the difference should not be significant. Sulfur levels in other areas of the world can be significantly higher for RM. Table 2-3, based on the ARB survey, provides the assumed mix of fuel types used for propulsion and auxiliary engines by ship type.

Table 2-3 Estimated Mix of Fuel Types Used by Ships

Ship Type	Fuel Used	
	Propulsion	**Auxiliary**
Passenger	100% RM	92% RM/8% MDO
Other	100% RM	71% RM/29% MDO

2.3.2.3.6 Propulsion and Auxiliary Engine Emission Factors

An analysis of emission data was prepared and published in 2002 by Entec.[17] The resulting Entec emission factors include individual factors for three speeds of diesel engines (slow-speed diesel (SSD), medium-speed diesel (MSD), and high-speed diesel (HSD)), steam turbines (ST), gas turbines (GT), and two types of fuel used here (RM and MDO). Table 2-4 lists the propulsion engine emission factors for NO_X and HC that were used for the 2002 port inventory development. The CO, PM, SO_2 and CO_2 emission factors shown in the table come from other data sources as explained below.

Table 2-4 Emission Factors for OGV Main Engines using RM, g/kWh

Engine	All Ports				West Coast Ports			Other Ports		
	NO_X	CO	HC	CO_2	PM_{10}	$PM_{2.5}$	SO_2	PM_{10}	$PM_{2.5}$	SO_2
SSD	18.1	1.40	0.60	620.62	1.4	1.3	9.53	1.4	1.3	10.29
MSD	14.0	1.10	0.50	668.36	1.4	1.3	10.26	1.4	1.3	11.09
ST	2.1	0.20	0.10	970.71	1.4	1.3	14.91	1.5	1.4	16.10
GT	6.1	0.20	0.10	970.71	1.4	1.3	14.91	1.5	1.4	16.10

CO emission factors were developed from information provided in the Entec appendices because they are not explicitly stated in the text. HC and CO emission factors were confirmed with a recent U.S. Government review.[18]

PM_{10}[A] values were determined based on existing engine test data in consultation with ARB.[19] GT PM_{10} emission factors were not part of the U.S. Government analysis but assumed here to be equivalent to ST PM_{10} emission factors. Test data shows PM_{10} emission rates as dependent upon fuel sulfur levels, with base PM_{10} emission rates of 0.23 g/kw-hr with distillate fuel (0.24% sulfur) and 1.35 g/kw-hr with residual fuel (2.46% sulfur).[20] The equation used to generate emission factors based on sulfur content is shown below. $PM_{2.5}$ is assumed to be 92 percent of PM_{10}. While the US Government NONROAD model uses 0.97 for such conversion based upon low sulfur fuels, a reasonable value seems to be closer to 0.92 because higher sulfur fuels in medium and slow speed engines would tend to produce larger particulates than high speed engines on low sulfur fuels.

Equation 2-2 Calculation of PM_{10} Emission Factors Based on Fuel Sulfur Levels

$$PM_{EF} = PM_{Nom} + [(S_{Act} - S_{Nom}) \times BSFC \times FSC \times MWR \times 0.0001]$$

where:

PM_{EF}	= PM emission factor adjusted for fuel sulfur
PM_{Nom}	= PM emission rate at nominal fuel sulfur level
	= 0.23 g/kW-hr for distillate fuel, 1.35 g/kW-hr for residual fuel
S_{Act}	= Actual fuel sulfur level (weight percent)
S_{Nom}	= nominal fuel sulfur level (weight percent)
	= 0.24 for distillate fuel, 2.46 for residual fuel
BSFC	= fuel consumption in g/kW-hr
	= 200 g/kW-hr used for this analysis
FSC	= percentage of sulfur in fuel that is converted to direct sulfate PM
	= 2.247% used for this analysis
MWR	= molecular weight ratio of sulfate PM to sulfur
	= 224/32 = 7 used for this analysis

SO_2 emission factors were based upon a fuel sulfur to SO_2 conversion formula which was supplied by ENVIRON.[21] Emission factors for SO_2 emissions were calculated using the formula assuming that 97.753 percent of the fuel sulfur was converted to SO_2.[22] The brake specific fuel consumption (BSFC)[B] that was used for SSDs was 195 g/kWh, while the BSFC that was used for MSDs was 210 g/kWh based upon Lloyds 1995. The BSFC that was used for STs and GTs was 305 g/kWh based upon Entec.[17]

Equation 2-3 Calculation of SO_2 Emission Factors, g/kWh

$$SO_2 \text{ EF} = \text{BSFC x } 64/32 \text{ x } 0.97753 \text{ x Fuel Sulfur Fraction}$$

CO_2 emission factors were calculated from the BSFC assuming a fuel carbon content of 86.7 percent by weight[17] and a ratio of molecular weights of CO_2 and C at 3.667.

Equation 2-4 Calculation of CO_2 Emission Factors, g/kWh

$$CO_2 \text{ EF} = \text{BSFC x } 3.667 \text{ x } 0.867$$

[A] PM_{10} is particulate matter of 10 micrometers or less.
[B] Brake specific fuel consumption is sometimes called specific fuel oil consumption (SFOC).

The most current set of auxiliary engine emission factors also comes from Entec except as noted below for PM and SO_2. Table 2-5 provides these auxiliary engine emission factors.

Table 2-5 Auxiliary Engine Emission Factors by Fuel Type, g/kWh

Engine	Fuel	All Ports				West Coast Ports			Other Ports		
		NO_X	CO	HC	CO_2	PM_{10}	$PM_{2.5}$	SO_2	PM_{10}	$PM_{2.5}$	SO_2
MSD	RM	14.70	1.10	0.40	668.36	1.4	1.3	10.26	1.4	1.3	11.09
	MDO	13.90	1.10	0.40	668.36	0.6	0.55	6.16	0.6	0.55	6.16

SO_2 emission factors were calculated using Equation 2-3 while PM emissions were determined using Equation 2-2.

Using the ratios of RM versus MDO use[15] as given in Table 2-3 together with the emission factors shown in Table 2-5, the auxiliary engine emission factor averages by ship type are listed in Table 2-6. As discussed above, this fuel sulfur level may be too high for the U.S. However, we do not believe this emission factor has a significant effect on the total emission inventory estimates.

Table 2-6 Auxiliary Engine Emission Factors by Ship Type, g/kWh

Ship Type	All Ports				West Coast Ports			Other Ports		
	NO_x	CO	HC	CO_2	PM_{10}	$PM_{2.5}$	SO_2	PM_{10}	$PM_{2.5}$	SO_2
Passenger	14.64	1.10	0.40	668.36	1.3	1.2	9.93	1.4	1.3	10.70
Others	14.47	1.10	0.40	668.36	1.1	1.0	9.07	1.2	1.1	9.66

2.3.2.3.7 Low Load Adjustment Factors for Propulsion Engines

Emission factors are considered to be constant down to about 20 percent load. Below that threshold, emission factors tend to increase as the load decreases. This trend results because diesel engines are less efficient at low loads and the brake specific fuel consumption (BSFC) tends to increase. Thus, while mass emissions (grams per hour) decrease with low loads, the engine power tends to decrease more quickly, thereby increasing the emission factor (grams per engine power) as load decreases. Energy and Environmental Analysis Inc. (EEA) demonstrated this effect in a study prepared for the U.S. Government in 2000.[23] In the EEA report, equations have been developed for the various emissions. The low-load emission factor adjustment factors were developed based upon the concept that the BSFC increases as load decreases below about 20 percent load.

Using these algorithms, fuel consumption and emission factors versus load were calculated. By normalizing emission factors to 20% load, low-load multiplicative adjustment factors were calculated for propulsion engines and presented in Table 2-7. Due to how they are operated, there is no need for a low load adjustment factor for auxiliary engines.

Table 2-7 Calculated Low Load Multiplicative Adjustment Factors

Load (%)	NO_X	HC	CO	PM	SO_2	CO_2
1	11.47	59.28	19.32	19.17	5.99	5.82
2	4.63	21.18	9.68	7.29	3.36	3.28
3	2.92	11.68	6.46	4.33	2.49	2.44
4	2.21	7.71	4.86	3.09	2.05	2.01
5	1.83	5.61	3.89	2.44	1.79	1.76
6	1.60	4.35	3.25	2.04	1.61	1.59
7	1.45	3.52	2.79	1.79	1.49	1.47
8	1.35	2.95	2.45	1.61	1.39	1.38
9	1.27	2.52	2.18	1.48	1.32	1.31
10	1.22	2.20	1.96	1.38	1.26	1.25
11	1.17	1.96	1.79	1.30	1.21	1.21
12	1.14	1.76	1.64	1.24	1.18	1.17
13	1.11	1.60	1.52	1.19	1.14	1.14
14	1.08	1.47	1.41	1.15	1.11	1.11
15	1.06	1.36	1.32	1.11	1.09	1.08
16	1.05	1.26	1.24	1.08	1.07	1.06
17	1.03	1.18	1.17	1.06	1.05	1.04
18	1.02	1.11	1.11	1.04	1.03	1.03
19	1.01	1.05	1.05	1.02	1.01	1.01
20	1.00	1.00	1.00	1.00	1.00	1.00

2.3.2.3.8 Use of Detailed Typical Port Data for Other Inputs

There is currently not enough information to readily calculate time-in-mode (hours/call) for all 117 ports during the maneuvering and hotelling modes of operation. As a result, it was necessary to review and select available detailed emission inventories that have been estimated for selected ports to date. These ports are referred to as typical ports. The typical port information for maneuvering and hotelling time-in-mode (as well as maneuvering load factors for the propulsion engines) was then used for the typical ports and also assigned to the other modeled ports. A modeled port is the port in which emissions are to be estimated. The methodology that was used to select the typical ports and match these ports to the other modeled ports is briefly described in Appendix 2D, and more fully described in an ICF report.[2]

2.3.2.3.9 Port Domestic Traffic

One of the concerns with using USACE entrances and clearances data is that it only contains foreign cargo movements moved by either a foreign flag vessel or a U.S. flag vessel. As a result, U.S. flag ships carrying domestic cargo (i.e., Jones Act) ships are not included in the USACE data. Determining the contribution of Jones Act ships is difficult as most data sources include Category 1 and 2 Jones Act ship movements with Category 3 ships and do not provide either enough data or a method for separating them.

Under contract to the U.S. Government, ICF conducted an analysis to estimate the amount of Category 3 Jones Act ships calling at the 117 U.S. ports. This was done by analyzing marine exchange data obtained from port authorities for eleven typical ports and using this information to estimate the Jones Act ship contribution for the remaining ports. Based on this limited analysis, Jones Act ships are estimated to account for 9.2% of the total installed power calling on U.S. ports. Approximately 30% of these ships, largely in the Alaska and Pacific regions, have been included in the 2002 baseline inventory. Based on this analysis, Jones Act ships excluded from this inventory constitute roughly 6.5% of total installed power.[24] This results in an underestimation of the port ship inventory and therefore the benefits of the ECA program reported in this chapter are also underestimated.

2.3.2.4 2002 Near Port Inventories

This section provides a summary of the total port emissions for 2002. Table 2-8 provides a breakout of the total port emissions by auxiliary and propulsion engines. Table 2-9 provides the breakout by mode of operation, while Table 2-10 provides a summary of port emissions by ship type.

Table 2-8 2002 Port Emissions Summary by Engine and Port Type (metric tonnes)

Engine Type	Port Type	Metric Tonnes						
		NO_X	PM_{10}	$PM_{2.5}$	HC	CO	SO_2	CO_2
Propulsion	Deep Water	64,288	5,478	5,034	2,532	6,329	52,676	2,360,435
	Great Lakes	248	25	23	11	22	187	11,267
	Total	64,536	5,503	5,057	2,543	6,351	52,863	2,371,702
Auxiliary	Deep Water	57,317	5,052	4,597	1,615	4,306	41,232	2,635,436
	Great Lakes	302	25	23	8	23	202	13,944
	Total	57,619	5,077	4,620	1,624	4,328	41,433	2,649,380
All	Deep Water	121,606	10,530	9,631	4,148	10,635	93,908	4,995,871
	Great Lakes	549	50	46	19	45	389	25,210
	Grand Total	122,155	10,580	9,677	4,167	10,680	94,297	5,021,082

Auxiliary emissions at ports are responsible for 39-48% of the total port inventory, depending on the pollutant. Hotelling, cruise, and RSZ modes of operation are all important contributors to emissions.

Table 2-9 2002 Port Emissions Summary by Mode and Port Type (metric tonnes)

Mode	Port Type	Metric Tonnes						
		NO_X	PM_{10}	$PM_{2.5}$	HC	CO	SO_2	CO_2
Cruise	Deep Water	34,193	2,826	2,623	1,141	2,651	21,186	1,314,146
	Great Lakes	183	17	16	6	14	137	8,313
	Total	34,376	2,843	2,639	1,148	2,665	21,323	1,322,459
RSZ	Deep Water	34,427	2,887	2,657	1,280	3,804	35,148	1,318,897
	Great Lakes	45	4	4	2	4	33	2,052
	Total	34,472	2,891	2,661	1,281	3,808	35,181	1,320,950
Maneuvering	Deep Water	7,383	758	625	440	724	4,356	266,262
	Great Lakes	70	7	7	4	8	50	3,213
	Total	7,452	765	632	444	732	4,406	269,476
Hotelling	Deep Water	45,603	4,060	3,726	1,287	3,456	33,218	2,096,566
	Great Lakes	252	21	19	7	19	168	11,631
	Total	45,855	4,081	3,745	1,294	3,475	33,386	2,108,197
All	Deep Water	121,606	10,530	9,631	4,148	10,635	93,908	4,995,871
	Great Lakes	549	50	46	19	45	389	25,210
	Grand Total	122,155	10,580	9,677	4,167	10,680	94,297	5,021,082

Table 2-10 2002 Port Emissions Summary by Ship Type and Port Type (metric tonnes)

Ship Type	Port Type	Metric Tonnes						
		NO_X	PM_{10}	$PM_{2.5}$	HC	CO	SO_2	CO_2
Auto Carrier	Deep Water	5,125	421	384	185	577	3,676	198,637
	Great Lakes	0	0	0	0	0	0	0
	Total	5,125	421	384	185	577	3,676	198,637
Barge Carrier	Deep Water	148	13	12	5	12	141	6,364
	Great Lakes	0	0	0	0	0	0	0
	Total	148	13	12	5	12	141	6,364
Self-Unloading Bulk Carrier	Deep Water	0	0	0	0	0	0	0
	Great Lakes	276	27	25	10	23	210	13,273
	Total	276	27	25	10	23	210	13,273
Other Bulk Carrier	Deep Water	19,373	1,570	1,431	633	1,732	14,945	767,825
	Great Lakes	227	19	17	7	18	147	9,807
	Total	19,600	1,589	1,448	640	1,750	15,092	777,632
Container	Deep Water	33,990	2,733	2,494	1,282	2,833	22,628	1,288,596
	Great Lakes	0	0	0	0	0	0	0
	Total	33,990	2,733	2,494	1,282	2,833	22,628	1,288,596
General Cargo	Deep Water	7,402	630	576	251	684	6,208	302,338
	Great Lakes	22	2	2	1	2	15	969
	Total	7,424	631	578	252	686	6,223	303,307
Miscellaneous	Deep Water	179	16	15	6	35	128	8,209
	Great Lakes	0	0	0	0	0	0	0

Ship Type	Port Type	Metric Tonnes						
		NO_X	PM_{10}	$PM_{2.5}$	HC	CO	SO_2	CO_2
	Total	179	16	15	6	35	128	8,209
Passenger	Deep Water	19,165	1,819	1,668	578	1,470	14,184	893,157
	Great Lakes	0	0	0	0	0	0	0
	Total	19,165	1,819	1,668	578	1,470	14,184	893,157
Refrigerated Cargo	Deep Water	3,027	247	226	98	313	1,968	130,060
	Great Lakes	0	0	0	0	0	0	0
	Total	3,027	247	226	98	313	1,968	130,060
Roll-On/Roll-Off	Deep Water	3,391	281	259	113	278	2,193	139,396
	Great Lakes	0	0	0	0	0	0	0
	Total	3,391	281	259	113	278	2,193	139,396
Tanker	Deep Water	29,758	2,796	2,562	994	2,695	27,802	1,259,107
	Great Lakes	22	2	2	1	2	15	1,012
	Total	29,780	2,798	2,564	995	2,697	27,817	1,260,119
Ocean Going Tug	Deep Water	48	5	4	2	6	34	2,182
	Great Lakes	0	0	0	0	0	0	0
	Total	48	5	4	2	6	34	2,182
Integrated Tug-Barge	Deep Water	0	0	0	0	0	0	0
	Great Lakes	3	0	0	0	0	2	149
	Total	3	0	0	0	0	2	149
All	Deep Water	121,606	10,530	9,631	4,148	10,635	93,908	4,995,871
	Great Lakes	549	50	46	19	45	389	25,210
	Grand Total	122,155	10,580	9,677	4,167	10,680	94,297	5,021,082

2.3.3 Interport Emissions

This section presents our nationwide analysis of the methodology and inputs used to estimate interport emissions from main propulsion and auxiliary engines used by Category 3 ocean-going vessels for the 2002 calendar year. The modeling domain for vessels operating in the ocean extends from the U.S. coastline to a 200 nautical mile boundary. For ships operating in the Great Lakes, it extends out to the international boundary with Canada. The emission results are divided into nine geographic regions of the U.S. (including Alaska and Hawaii), and then totaled to provide a national inventory.

The interport emissions described in this section represent total interport emissions prior to any adjustments made to incorporate near-port inventories. The approach used to replace the near-port portion of the interport emissions is provided in Section 2.3.4.

2.3.3.1 Interport Methodology

The interport emissions were estimated using the Waterway Network Ship Traffic, Energy, and Environmental Model (STEEM).[3,4] STEEM was developed by the University of Delaware as a comprehensive approach to quantify and geographically represent interport ship traffic, emissions, and energy consumption from large vessels calling on U.S. ports or transiting the U.S. coastline to other destinations, and shipping activity in Canada and Mexico. The model estimates emissions from main propulsion and auxiliary marine engines used on Category 3 vessels that engage in foreign commerce using historical North American shipping activity, ship attributes (i.e., characteristics), and activity-based emission factor information. These inputs are assembled using a GIS platform that also contains an empirically derived network of shipping lanes. It includes the emissions for all ship operational modes from cruise in unconstrained shipping lanes to maneuvering in a port. The model, however, excludes hotelling operations while the vessel is docked or anchored, and very low speed maneuvering close to a dock. For that reason, STEEM is referred to as an "interport" model, to easily distinguish it from the near ports analysis.

STEEM uses advanced ArcGIS tools and develops emission inventories in the following way. The model begins by building a spatially-defined waterway network based on empirical shipping location information from two global ship reporting databases. The first is the International Comprehensive Ocean-Atmosphere Data Set (ICOADS), which contains reports on marine surface and atmospheric conditions from the Voluntary Observing Ships (VOS) fleet.[25] There are approximately 4,000 vessels worldwide in the VOS system. The ICOADS project is sponsored by the National Oceanic and Atmospheric Administration and National Science Foundation's National Center for Atmospheric Research (NCAR). The second database is the Automated Mutual-Assistance Vessel Rescue (AMVER) system.[26] The AMVER data set is based on a ship search and rescue reporting network sponsored by the U.S. Coast Guard. The AMVER system is also voluntary, but is generally limited to ships over 1,000 gross tons on voyages of 24 hours or longer. About 8,600 vessels reported to AMVER in 2004.

The latitude and longitude coordinates for the ship reports in the above databases are used to statistically create and spatially define the direction and width of each shipping lane in the waterway network. Each statistical lane (route and segment) is given a unique identification number for computational purposes. For the current analysis, STEEM used 20 years of ICOADS data (1983-2002) and about one year of AMVER data (part of 2004 and part of 2005) (Figure 2-2).

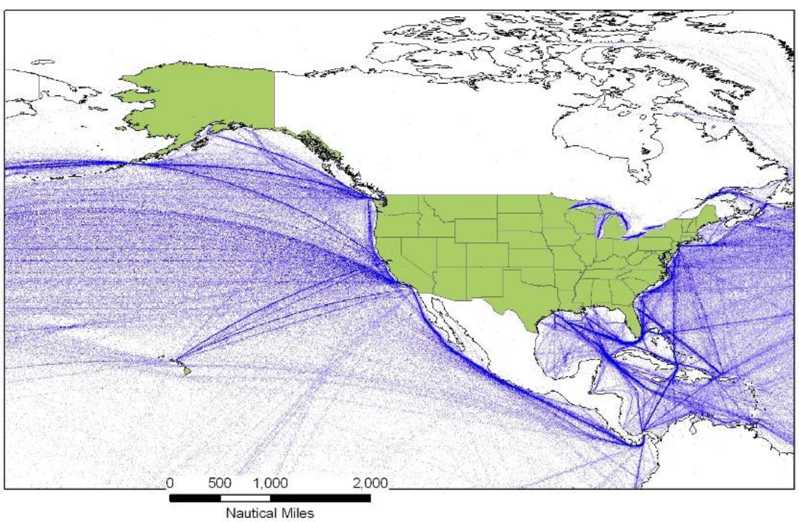


Figure 2-2 AMVER and ICOADS data

Every major ocean and Great Lake port is also spatially located in the waterway network using ArcGIS software. For the U.S., the latitude and longitude for each port is taken from the USACE report on vessel entrances and clearances.[11] Each port also has a unique identification number for computational purposes.

As illustrated in Figure 2-3, the waterway network represented by STEEM resembles a highway network on land. It is composed of ports, which are origins and destinations of shipping routes: junctions where shipping routes intersect, and segments that are shipping lanes between two connected junctions. Each segment can have only two junctions or ports, and ship traffic flow can enter and leave a segment only through a junction or at a port. The figure represents only a sample of the many routes contained in the model.

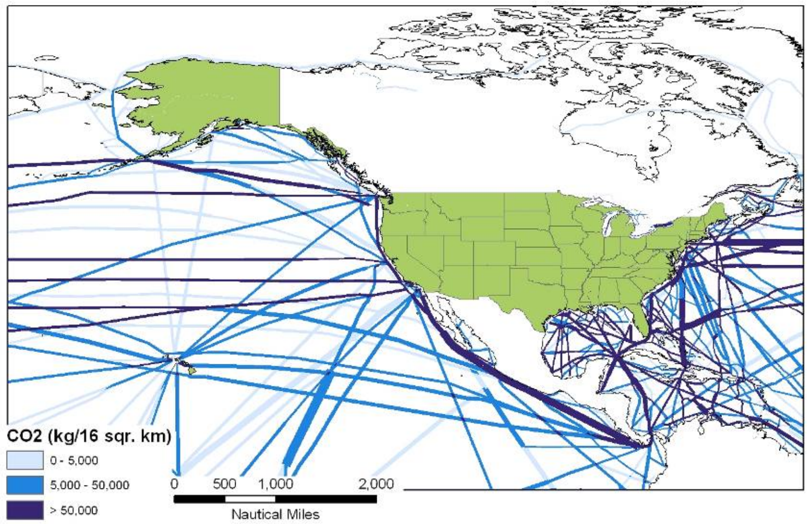


Figure 2-3 Illustration of STEEM Modeling Domain and Spatial Distribution of Shipping Lanes

The STEEM interport model also employs a number of databases to identify the movements for each vessel (e.g., trips), individual ship attributes (e.g., vessel size and horsepower), and related emission factor information (e.g., emission rates) that are subsequently used in the inventory calculations.

To allocate ships to the statistical lanes, STEEM uses ArcGIS Network Analyst tools along with specific information on each individual ship movement to solve the most probable path on the network between each pair of ports (i.e., a trip) for a certain ship size. This is assumed to represent the least-energy path, which in most cases is the shortest distance unless prevented by weather or sea conditions, water depth, channel width, navigational regulations, or other constraints that are beyond the model's capability to forecast.

After identifying the shipping route and resulting distance associated with each unique trip, the emissions are simply calculated for each operational mode using the following generalized equation along with information from the ship attributes and emission factor databases:

Equation 2-5

Emissions per trip = distance (nautical miles) / speed (nautical miles/hour) x horsepower (kW) x fractional load factor x emission factor (g/kW-hour)

In STEEM, emissions are calculated separately for distances representing cruise and maneuvering operational modes. Maneuvering occurs at slower speeds and load factors than during cruise conditions. In STEEM, maneuvering is assumed to occur for the first and last

20 kilometers of each trip when a ship is entering or leaving a port. A ship is assumed to move at maneuvering speed for an entire trip if the distance is less than 20 kilometers.

Finally, the emissions along each shipping route (i.e., segment) for all trips are proportioned among the respective cells that are represented by the gridded modeling domain. For this work, emissions estimates were produced at a cell resolution of 4 kilometers by 4 kilometers, which is appropriate for most atmospheric air quality models. The results for each cell are then summed, as appropriate, to produce emission inventories for the various geographic regions of interest in this analysis.

2.3.3.2 Inputs for Interport Emission Calculations

The STEEM model includes detailed information about ship routes and destinations in order to provide spatially allocated emissions of ships in transit. The shipping lanes and directions were empirically derived from ship positioning data in several datasets. The International Comprehensive Ocean-Atmosphere Data Set (ICOADS) contains reports on marine surface and atmospheric conditions from the Voluntary Observing Ships (VOS) fleet.[27] STEEM also uses a dataset derived from the Automated Mutual-Assistance Vessel Rescue (AMVER) system,[28] which is based on a ship search and rescue reporting network sponsored by the U.S. Coast Guard. Traffic along each of these lanes is derived from USACE entrance and clearance data for 2002,[29] together with Lloyd's Register-Fairplay Ltd's data for ship characteristics. Information for number of calls, ship characteristics, propulsion engine power, and cruise speed were obtained from these data.

The emission factors and load factors used as inputs to STEEM are very similar to those used for the ports analysis. Additional adjustments were made to interport emission results for PM_{10} and SO_2 in order to reflect recent U.S. Government review of available engine test data and fuel sulfur levels. Details of the STEEM emission inputs and adjustments are located in Appendix 2E.

2.3.3.3 Interport Domestic Traffic

As previously noted, STEEM includes the emissions associated with ships that are engaged in foreign commerce. As a result, U.S. flag vessels carrying domestic cargo (Jones Act ships) are not included. The STEEM interport analysis also roughly estimated the emissions associated with these ships that are engaged solely in domestic commerce.[1,4] Specifically, the interport analysis estimated that the large ocean-going vessels carrying only domestic cargo excluded from STEEM represent approximately 2-3 percent of the total U.S. emissions.

In section 2.3.2.3.9 in the estimation of port inventories, the estimate of excluded installed power was roughly 6.5 percent. It is not inconsistent that the STEEM estimate of excluded emissions is lower than the excluded power estimated from calls to U.S. ports, since the STEEM model includes ships that are transiting without stopping at U.S. ports. Since most of the Jones Act ships tend to travel closer along the coast line, most of the Jones Act ship traffic is expected to fall within the proposed ECA. Therefore, the results presented in this chapter are expected to underpredict the benefits of the proposed ECA.

2.3.4 Combining the Near Port and Interport Inventories

The national and regional inventories in this study are a combination of the results from the near ports analysis described in Section 2.3.2 and the STEEM interport modeling described in Section 2.3.3. The two inventories are quite different in form. As previously presented, the STEEM modeling domain spans the Atlantic and Pacific Oceans in the northern hemisphere. The model characterizes emissions from vessels while traveling between ports. That includes when a vessel is maneuvering a distance of 20 kilometers to enter or exit a port, cruising near a port as it traverses the area, or moving in a shipping lane across the open sea. For the U.S., STEEM includes the emissions associated with 251 ports. The results are spatially reported in a gridded format that is resolved to a cell dimension of 4 kilometers by 4 kilometers.

The near port results, however, are much more geographically limited and are not reported in a gridded format. The analysis includes the emissions associated with ship movements when entering or exiting each of 117 major U.S. ports. For deep water ports that include when a vessel is hotelling and maneuvering in the port, operating in the RSZ that varies in length for each port, and cruising 25 nautical miles between the end of the RSZ and an unconstrained shipping lane. For Great Lakes ports that includes hotelling and maneuvering, three nautical miles of RSZ operation, and cruising 7 nautical miles between the end of the RSZ and open water. The results are reported for each port and mode of operation.

To precisely replace only the portion of the STEEM interport inventory that is represented in the near port inventory results, it is necessary to spatially allocate the emissions in a format that is compatible with the STEEM 4 kilometers by 4 kilometers gridded output. Once that has been accomplished, the two inventories can be blended together. Both of these processes are described below. This work was conducted by ENVIRON International as a subcontractor under the U.S. Government contract with ICF.[2]

2.3.4.1 Spatial Location of the Near Port Inventories

The hotelling, maneuvering, RSZ, and cruise emissions from the near port inventories were spatially located by their respective latitude and longitude coordinates using ArcGIS software. For this study, shapefiles were created that depicted the emission locations as described above. Additional shapefiles were also obtained to locate other geographic features such as the coastline and rivers of the U.S. These shapefiles and the STEEM output can be layered upon each other, viewed in ArcMap, and analyzed together. The following sections provide a more detailed description of how the shapefiles representing the ports, RSZ lanes, and cruise lanes were developed.

2.3.4.1.1 Ports

Each port, and thus the designated location for hotelling and maneuvering emissions, is modeled as a single latitude/longitude coordinate point using the port center as defined by USACE in the Principal Ports of the United States dataset.[10] The hotelling and maneuvering emissions represented by the latitude/longitude coordinate for each port were subsequently assigned to a single cell in the gridded inventory where that point was located. It should be noted that modeling a port as a point will over specify the location of the emissions associated with that port if it occupies an area greater than one grid cell, or 4 kilometers by 4 kilometers. The coordinates of all of the 117 ports used in this work are shown in Appendix 2A.

2.3.4.2 Reduced Speed Zone Operation

The RSZ routes associated with each of the 117 ports were modeled as lines. Line shapefiles were constructed using the RSZ distance information described in Section 2.3.2 and the USACE National Waterway Network (NWN) geographic database of navigable waterways in and around the U.S.[14] The coordinates of RSZ endpoints for all of the 117 ports used in this work are shown in Appendix 2C.

The RSZ emissions were distributed evenly along the length of the line. The latitude/longitude coordinates for each point along the line were subsequently used to assign the emissions to a grid cell based on the proportion of the line segment that occurred in the respective cell.

2.3.4.2.1 Cruise Operations

The cruise mode links that extend 25 nautical miles for deep water ports or 7 nautical miles for Great Lake ports from the end of the RSZ end point were also modeled with line shapefiles. These links were spatially described for each port following the direction of the shipping lane evident in the STEEM data. Again, as with RSZ emissions, the latitude/longitude coordinates for each point along the line were subsequently used to assign the emissions to a grid cell based on the proportion of the line segment that occurred in the respective cell.

2.3.4.3 Combining the Near Port and STEEM Emission Inventories in Port Areas

After spatially defining the geographic location of the near port emissions, but before actually inserting them into the gridded STEEM inventory, it was necessary to determine if all of the STEEM emissions within an affected cell should be replaced, or if some of the emissions should be retained. In this latter case, ships would be traversing the area near a port, but not actually entering or exiting the port.

The percentage of STEEM emissions that are attributable to a port, and should be removed and replaced, were approximated by dividing the STEEM emissions in the isolated portion of the route that lead only to the port, with the STEEM emissions in the major shipping lane.

The actual merging of the two inventories was performed by creating a number of databases that identified the fraction of the near port inventory for each pollutant species and operating mode that should be added to the grid cells for each port. A similar database was also created that identified how much of the original STEEM emissions should be reduced to account for ship movements associated directly with a port, while preserving those that represented transient vessel traffic. These databases were subsequently used to calculate the new emission results for each affected cell in the original STEEM gridded inventory, resulting in the combined inventory results for this study.

In a few cases, the outer edges of the port inventories fell outside the international boundary; that portion outside the U.S. boundary was removed. As a result, the port totals presented in the next section are slightly less than those reported in Section 2.3.2.4. The removed portion represents less than 2 percent of the total port emissions.

Since STEEM includes emissions associated with 251 ports, the 117 ports do not cover all the ports identified by the shipping lane paths evident in the STEEM data. In the remaining ports, the STEEM model output was used.

2.3.5 2002 Baseline Inventories

The modeling domain of the new combined emission inventory described above is the same as the original STEEM domain, i.e., the Atlantic and Pacific Oceans, the Gulf of Mexico, the Great Lakes, Alaska, and Hawaii. Inventories for the nine geographic regions of the U.S. specified in Section 2.2 were created using ArcGIS software to intersect the regional shapefiles with the 4 kilometers by 4 kilometers gridded domain. Any grid cell split by a regional boundary was considered to be within a region if over 50 percent of its area was within the region. The emissions from the cells within a region were then summed. The final emission inventories for 2002 are shown in Table 2-11 for each of the nine geographic regions and the nation. The geographic scope of these regions was previously displayed in Figure 2-1.

Table 2-11 2002 Regional and National Emissions from Category 3 Vessel Main and Auxiliary Engines

Region	Metric Tonnes						
	NO_X	PM_{10}	$PM_{2.5}$ [a]	HC	CO	SO_2	CO_2
Alaska East (AE)	18,051	1,425	1,311	597	1,410	10,618	657,647
Alaska West (AW)	60,019	4,689	4,313	1,989	4,685	34,786	2,143,720
East Coast (EC)	219,560	17,501	16,101	7,277	17,231	145,024	8,131,553
Gulf Coast (GC)	172,897	14,043	12,920	5,757	14,169	104,852	6,342,139
Hawaii East (HE)	22,600	1,775	1,633	749	1,765	13,182	818,571
Hawaii West (HW)	31,799	2,498	2,297	1,053	2,484	18,546	1,151,725
North Pacific (NP)	26,037	2,154	1,982	938	2,090	15,295	990,342
South Pacific (SP)	104,155	8,094	7,447	3,464	8,437	60,443	3,796,572
Great Lakes (GL)	15,019	1,179	1,085	498	1,174	8,766	541,336
Total Metric Tonnes	*670,135*	*53,358*	*49,089*	*22,324*	*53,444*	*411,511*	*24,573,605*

[a] Estimated from PM_{10} using a multiplicative adjustment factor of 0.92.

The relative contributions of the near port and interport emission inventories to the total U.S. ship emissions are presented in Table 2-12 and Table 2-13. As expected, based on the geographic scope of the two types of inventories, the interport and near port inventories are about 80 percent and 20 percent of the total, respectively.

Table 2-12 2002 Contribution of Near Port and Interport Emissions to the Total C3 Inventory

Region	Metric Tonnes								
	NO_X			PM_{10}			$PM_{2.5}$ [a]		
	Port	Interport	Total	Port	Interport	Total	Port	Interport	*Total*
Alaska East (AE)	833	17,218	18,051	80	1,345	1,425	74	1,237	*1,311*
Alaska West (AW)	0	60,019	60,019	0	4,689	4,689	0	4,313	*4,313*
East Coast (EC)	48,313	171,247	219,560	4,126	13,375	17,501	3,796	12,305	*16,101*
Gulf Coast (GC)	33,637	139,260	172,897	3,169	10,874	14,043	2,916	10,004	*12,920*
Hawaii East (HE)	2,916	19,684	22,600	251	1,524	1,775	231	1,402	*1,633*
Hawaii West (HW)	0	31,799	31,799	0	2,498	2,498	0	2,297	*2,297*
North Pacific (NP)	14,015	12,022	26,037	1,216	938	2,154	1,119	863	*1,982*
South Pacific (SP)	20,079	84,076	104,155	1,525	6,569	8,094	1,403	6,044	*7,447*
Great Lakes (GL)	491	14,528	15,019	44	1,135	1,179	41	1,044	*1,085*
Total Metric Tonnes	*120,285*	*549,852*	*670,137*	*10,413*	*42,945*	*53,358*	*9,580*	*39,510*	*49,089*

[a] Estimated from PM_{10} using a multiplicative adjustment factor of 0.92.

Table 2-13 2002 Contribution of Near Port and Interport Emissions to the Total C3 Inventory (Cont'd)

Region	Metric Tonnes								
	HC			CO			SO_2		
	Port	Interport	Total	Port	Interport	Total	Port	Interport	*Total*
Alaska East (AE)	27	570	597	66	1,344	1,410	641	9,977	*10,618*
Alaska West (AW)	0	1,989	1,989	0	4,685	4,685	0	34,786	*34,786*
East Coast (EC)	1,603	5,674	7,277	3,864	13,367	17,231	45,952	99,072	*145,024*
Gulf Coast (GC)	1,142	4,615	5,757	3,305	10,864	14,169	24,187	80,665	*104,852*
Hawaii East (HE)	96	653	749	230	1,535	1,765	1,891	11,291	*13,182*
Hawaii West (HW)	0	1,053	1,053	0	2,484	2,484	0	18,546	*18,546*
North Pacific (NP)	540	398	938	1,152	938	2,090	8,329	6,966	*15,295*
South Pacific (SP)	678	2,786	3,464	1,876	6,561	8,437	11,715	48,728	*60,443*
Great Lakes (GL)	17	481	498	40	1,134	1,174	346	8,420	*8,766*
Total Metric Tonnes	*4,103*	*18,219*	*22,322*	*10,533*	*42,912*	*53,445*	*93,062*	*318,450*	*411,512*

As noted previously, these inventories exclude a portion of traffic from U.S. flag ships carrying domestic cargo. Estimates range from roughly 2 to 7 percent of installed power, which indicates that the inventories may be underestimated by 2 to 7 percent.

2.4 Development of 2020 Inventories

2.4.1 Outline of Methodology

The emissions from Category 3 ocean-going vessels (main propulsion and auxiliary engines) are projected to 2020 by applying certain growth factors to the 2002 emission inventories to account for the expected change in ship traffic over these time periods due to growth in trade.

The remaining sections describe the derivation of the growth adjustment factors for each of the modeling regions described in Section 2.2. Emission control program related adjustments to the

2020 inventories are then described. A baseline inventory and an inventory within the proposed ECA are then presented.

2.4.2 Growth Factors by Geographic Region

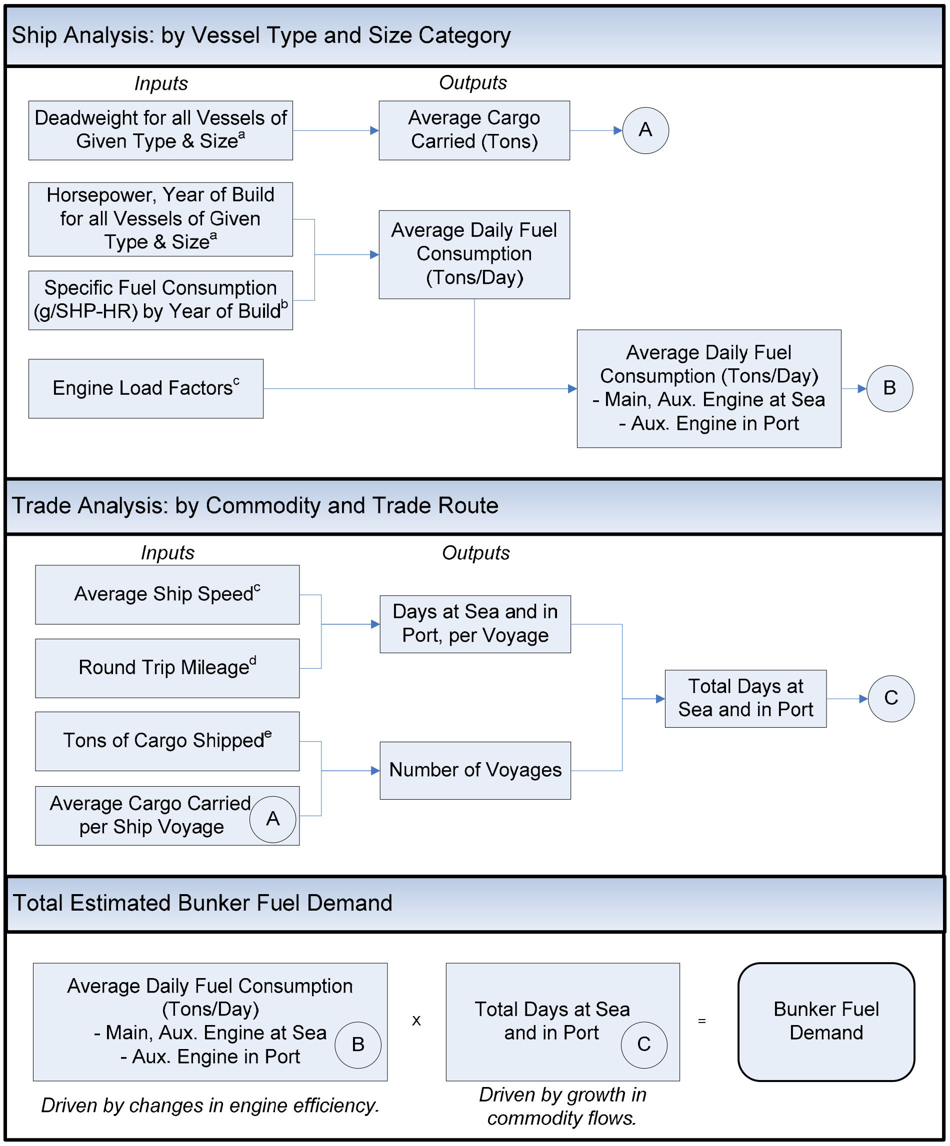
The growth factors that are used to estimate future year emission inventories are based on the expected demand for marine bunker fuels that is associated with shipping goods, i.e., commodities, into and out of the U.S. This section describes the growth factors that are used to project the emissions to 2020 for each of the nine geographic regions evaluated in this analysis. The use of bunker fuel as a surrogate for estimating future emissions is appropriate because the quantity of fuel consumed by C3 engines is highly correlated with the amount of combustion products, i.e., pollutants that are emitted from those vessels. The term bunker fuel in this report also includes marine distillate oil and marine gas oil that are used in some auxiliary power engines.

The remainder of this section first summarizes the development of growth rates by RTI International (RTI) for five geographic regions of the U.S., as performed under contract to the U.S. government.[5,6] This is followed by the derivation of the growth factors that are used in this study for the nine geographic regions of interest.

2.4.2.1 Summary of Regional Growth Rate Development

RTI developed fuel consumption growth rates for five geographic regions of the U.S. These regions are the East Coast, Gulf Coast, North Pacific, South Pacific, and Great Lakes. The amount of bunker fuel required in any region and year is based on the demand for transporting various types of cargo by Category 3 vessels. This transportation demand is in turn driven by the demand for commodities that are produced in one location and consumed in another, as predicted by an econometric model. The flow of commodities is matched with typical vessels for trade routes (characterized according to cargo capacity, engine horsepower, age, specific fuel consumption, and engine load factors). Typical voyage parameters are then assigned to the trade routes that include average ship speed, round trip mileage, tons of cargo shipped, and days in port. Fuel consumption for each trade route and commodity type thus depends on commodity projections, ship characteristics, and voyage characteristics. Figure 2-4 illustrates the approach to developing baseline projections of marine fuel consumption.

As a means of comparison, the IMO Secretary General's Informal Cross Government/Industry Scientific Group of Experts presented a growth rate that ranged from 3.3% to 3.7%.[30] RTI's overall U.S. growth rate was projected at 3.4%, which is consistent with that range.



a – Clarksons Ship Register Database
b – Engine Manufacturers' Data, Technical Papers
c – Corbett and Wang (2005) "Emission Inventory Review: SECA Inventory Progress Discussion"
d - Combined trade routes and heavy leg analysis
e – Global Insight Inc. (GII) Trade Flow Projections

Figure 2-4 Illustration of Method for Estimating Bunker Fuel Demand

2.4.2.2 Trade Analysis

The trade flows between geographic regions of the world, as illustrated by the middle portion of Figure 2-4 were defined for the following eight general types of commodities:

- liquid bulk – crude oil
- liquid bulk – refined petroleum products
- liquid bulk – residual petroleum products
- liquid bulk – chemicals (organic and inorganic)
- liquid bulk –gas (including LNG and LPG)
- dry bulk (e.g., grain, coal, steel, ores and scrap)
- general cargo (e.g., lumber/forest products)

- containerized cargo

The analysis specifically evaluated trade flows between 21 regions of the world. Table 2-14 shows the countries associated with each region.

Table 2-14 Aggregate Regions and Associated Countries

Aggregate Regions	Base Countries / Regions
U.S. Atlantic Coast	U.S. Atlantic Coast
U.S. Great Lakes	U.S. Great Lakes
U.S. Gulf Coast	U.S. Gulf Coast
E. Canada[a]	Canada[a]
W. Canada[a]	Canada[a]
U.S. Pacific North	U.S. Pacific North
U.S. Pacific South	U.S. Pacific South
Greater Caribbean	Colombia, Mexico, Venezuela, Caribbean Basin, Central America
South America	Argentina, Brazil, Chile, Peru, Other East Coast of S. America, Other West Coast of S. America
Africa – West	Western Africa
Africa-North/East-Mediterranean	Mediterranean Northern Africa, Egypt, Israel,
Africa-East/South	Kenya, Other Eastern Africa, South Africa, Other Southern Africa
Europe-North	Austria, Belgium, Denmark, Finland, France, Germany, Ireland, Netherlands, Norway, Sweden, Switzerland, United Kingdom
Europe-South	Greece, Italy, Portugal, Spain, Turkey, Other Europe
Europe-East	Bulgaria, Czech Republic, Hungary, Poland, Romania, Slovak Republic
Caspian Region	Southeast CIS
Russia/FSU	The Baltic States, Russia Federation, Other Western CIS
Middle East Gulf	Jordan, Saudi Arabia, UAE, Other Persian Gulf
Australia/NZ	Australia, New Zealand
Japan	Japan
Pacific-High Growth	Hong Kong S.A.R., Indonesia, Malaysia, Philippines, Singapore, South Korea, Taiwan, Thailand
China	China
Rest of Asia	Viet Nam, India, Pakistan, Other Indian Subcontinent

[a] Canada is treated as a single destination in the GI model. Shares of Canadian imports from and exports to regions of the world in 2004 are used to divide Canada trade into shipments to/from Eastern Canada ports and shipments to/from Western Canada ports.[31]

The overall forecast of demand for shipping services and bunker fuel was determined for each of the areas using information on commodity flows from Global Insight's (GI) World Trade Service. Specifically, GI provided a specialized forecast that reports the flow of each commodity type for the period 1995–2024, based on a proprietary econometric model. The general structure of the GI model for calculating trade flows assumes a country's imports from another country are driven by the importing country's demand forces (given that the exporting country possesses enough supply capacity), and affected by exporting the country's export price and importing country's import cost for the commodity. The model then estimates demand forces, country-specific exporting capacities, export prices, and import costs.

The GI model included detailed annual region-to-region trade flows for eight composite commodities from 1995 to 2024, in addition to the total trade represented by the commodities. Table 2-15 illustrates the projections for 2012 and 2020, along with baseline data for 2005. In 2005, dry bulk accounted for 41 percent of the total trade volume, crude oil accounted for 28 percent, and containers accounted for 12 percent. Dry bulk and crude oil shipments are expected to grow more slowly over the forecast period than container shipments. By 2020, dry bulk represents 39 percent of the total, crude oil is 26 percent, and containers rise to 17 percent.

Table 2-15 Illustration of World Trade Estimates for Composite Commodities, 2005, 2012, and 2020

Commodity Type	Cargo (millions of tons)		
	2005	2012	2020
Dry Bulk	2,473	3,051	3,453
Crude Oil	1,703	2,011	2,243
Container	714	1,048	1,517
Refined Petroleum	416	471	510
General Cargo	281	363	452
Residual Petroleum and Other Liquids	190	213	223
Chemicals	122	175	228
Natural Gas	79	91	105
Total International Cargo Demand	5,979	7,426	8,737

2.4.2.3 Ship Analysis by Vessel Type and Size

Different types of vessels are required to transport the different commodities to the various regions of the world. Profiles of these ships were developed to identify the various vessel types and size categories that are assigned to transport commodities of each type along each route. These profiles include attributes such as ship size, engine horsepower, engine load factors, age, and engine fuel efficiency. This information was subsequently used to estimate average daily fuel consumption for each typical ship type and size category.

The eight GI commodity categories were mapped to the type of vessel that would be used to transport that type of cargo using information from Clarkson's Shipping Database.[32] These assignments are shown in Table 2-16.

Table 2-16 Assignment of Commodities to Vessel Types

Commodity	Ship Category	Vessel Type
Liquid bulk – crude oil	Crude Oil Tankers	Tanker
Liquid bulk – refined petroleum products	Product Tankers	Product Carrier
Liquid bulk – residual petroleum products	Product Tankers	Product Carrier
Liquid bulk – chemicals (organic and inorganic)	Chemical Tankers	Chemical & Oil Carrier
Liquid bulk – natural gas (including LNG and LPG)	Gas Carriers	LNG Carrier, LPG Carrier, Chemical & LPG Carrier, Ethylene/LPG, Ethylene/LPG/Chemical, LNG/Ethylene/LPG, LNG/Regasification, LPG/Chemical, LPG/Oil, Oil & Liquid Gas Carrier
Dry bulk (e.g. grain, coal, steel, ores and scrap)	Dry Bulk Carriers	Bulk Carrier
General cargo (including neobulk, lumber/forest products)	General Cargo	General Cargo Liner, Reefer, General Cargo Tramp, Reefer Fish Carrier, Ro-Ro, Reefer/Container, Ro-Ro Freight/Passenger, Reefer/Fleet Replen., Ro-Ro/Container, Reefer/General Cargo, Ro-Ro/Lo-Lo, Reefer/Pallets Carrier, Reefer/Pass./Ro-Ro, Reefer/Ro-Ro Cargo
Containerizable cargo	Container Ships	Fully Cellular Container

Each of the vessel types were classified by their cargo carrying capacity or deadweight tons (DWT). The size categories were identified based on both industry definitions and natural size breaks within the data. Table 2-17 summarizes the size categories that were used in the analysis and provides other information on the general attributes of the vessels from Clarkson's Shipping Database. The vessel size descriptions are also used to define shipping routes based on physical limitations that are represented by canals or straits through which ships can pass. Very large crude oil tankers are the largest by DWT rating, and the biggest container ships (Suezmax) are also very large.

Table 2-17 Fleet Characteristics

Ship Type	Size by DWT	Maximum Size (DWT)	Maximum Size (DWT)	Number of Ships	Total DWT (millions)	Total Horse Power (millions)	Total Kilowatts (millions)
Container	Suezmax	83,000	140,000	101	9.83	8.56	6.38
	PostPanamax	56,500	83,000	465	30.96	29.3	21.85
	Panamax	42,100	56,500	375	18.04	15.04	11.21
	Intermediate	14,000	42,100	1,507	39.8	32.38	24.14
	Feeder	0	14,000	1,100	8.84	7.91	5.90
General Cargo	All	All		3,214	26.65	27.07	20.18
Dry Bulk	Capesize	79,000	0	715	114.22	13.81	10.30
	Panamax	54,000	79,000	1,287	90.17	16.71	12.46
	Handymax	40,000	54,000	991	46.5	10.69	7.97
	Handy	0	40,000	2,155	58.09	19.58	14.60
Crude Oil Tanker	VLCC	180,000	0	470	136.75	15.29	11.40
	Suezmax	120,000	180,000	268	40.63	5.82	4.34
	AFRAmax	75,000	120,000	511	51.83	8.58	6.40
	Panamax	43,000	75,000	164	10.32	2.17	1.62
	Handymax	27,000	43,000	100	3.45	1.13	0.84
	Coastal	0	27,000	377	3.85	1.98	1.48
Chemical Tanker	All	All		2,391	38.8	15.54	11.59
Petroleum Product Tanker	AFRAmax	68,000	0	226	19.94	3.6	2.68
	Panamax	40,000	68,000	352	16.92	4.19	3.12
	Handy	27,000	40,000	236	7.9	2.56	1.91
	Coastal	0	27,000	349	3.15	1.54	1.15
Natural Gas Carrier	VLGC	60,000	0	157	11.57	5.63	4.20
	LGC	35,000	60,000	140	6.88	2.55	1.90
	Midsize	0	35,000	863	4.79	3.74	2.79
Other	All	All		7,675	88.51	53.6	39.96
Total	--	--	--	26,189	888.4	308.96	230.36

The average fuel consumption for each vessel type and size category was estimated in a multi-step process using individual vessel data on engine characteristics. Clarkson's Shipping Database Register provides each ship's total installed horsepower (HP), type of propulsion (diesel or steam), and year of build. These characteristics are then matched to information on typical specific fuel consumption (SFC), which is expressed in terms of grams of bunker fuel burned per horsepower-hour (g/HP-hr, which is equivalent to 1.341 g/kW-hr).

The SFC values are based on historical data from Wartsila Sulzer, a popular manufacturer of diesel engines for marine vessels. RTI added an additional 10 percent to the reported "test bed" or "catalogue" numbers to account for the guaranteed tolerance level and an in-service SFC differential. Overall, the 10 percent estimate is consistent with other analyses that show some variation between the “test bed” SFC values reported in the manufacturer product catalogues and those observed in actual service. This difference is explained by the fact that old, used engines consume more fuel than brand new engines and in-service fuels may be different than the test bed fuels.[33]

Figure 2-5 shows SFC values that were used in the model regarding the evolution of specific fuel oil consumption rates for diesel engines over time. Engine efficiency in terms of SFC has improved over time, most noticeably in the early 1980s in response to rising fuel prices. However, there is a tradeoff between improving fuel efficiency and reducing emissions. Conversations with engine manufacturers indicate that it is reasonable to assume SFC will remain constant for the projection period of this study, particularly as they focus on meeting NO_X emission standard as required by MARPOL Annex VI, or other potential pollution control requirements. Post-2000 SFC values are constant at approximately 135 g/hp-hr (180 g/kW-hr).

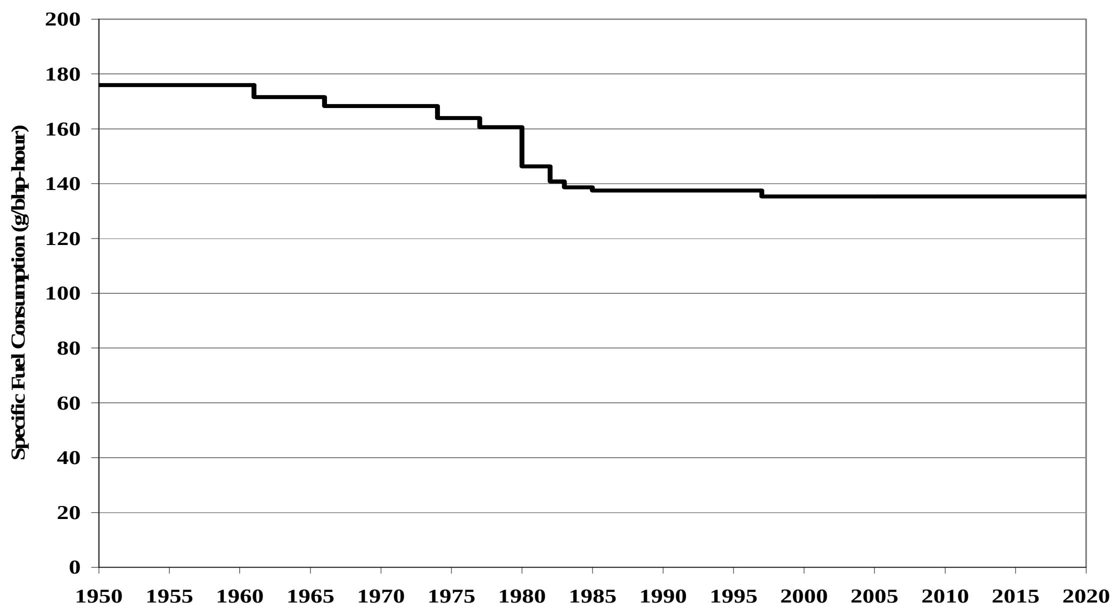


Figure 2-5 Diesel Engine Specific Fuel Consumption

RTI assumed a fixed SFC of 220 g/HP-hr (295 g/kW-hr) for steam engines operating on bunker fuel.

Using the above information, the average daily fuel consumption (AFC), expressed in metric tons of fuel at full engine load, for each vessel type and size category is found using the following equation:

Equation 2-6

$$\text{Fleet AFC}_{v,s} = \frac{1}{N}\sum\left[SFC_{v,s} \times HP_{v,s} \times 10^{-6} tonnes/g\right]$$

Where:

- Fleet AFC = Average daily fuel consumption in metric tonnes at full engine load
- *v* = Vessel type
- *s* = Vessel size category
- N = Number of vessels in the fleet
- SFC = Specific fuel consumption in grams of bunker fuel burned per horsepower-hour in use(g/HP-hr)
- HP = Total installed engine power, in horsepower (HP)
- 10^6 tonnes/g = Conversion from grams to metric tonnes

As previously noted, AFC values calculated in the above equation are based on total horsepower; therefore, they must be scaled down to reflect typical operation using less than 100 percent of the horsepower rating, i.e., actual engine load. Table 2-18 shows the engine load factors that were used to estimate the typical average daily fuel consumption (tons/day) for the main propulsion engine and the auxiliary engines when operated at sea and in port.[34]

Table 2-18 Main and Auxiliary Engine Load Factors

Vessel Type	Main Engine Load Factor (%)	Auxiliary Engine as Percent of Main Engine	Auxiliary Engine as Percent of Main Engine at Sea
Container Vessels	80	22.0	11.0
General Cargo Carriers	80	19.1	9.5
Dry Bulk Carriers	75	22.2	11.1
Crude Oil Tankers	75	21.1	10.6
Chemical Tankers	75	21.1	10.6
Petroleum Product Tankers	75	21.1	10.6
Natural Gas Carrier	75	21.1	10.6
Other	70	20.0	10.0

The RTI analysis also assumed that the shipping fleet changes over time as older vessels are scrapped and replaced with newer ships. Specifically, vessels over 25 years of age are retired and replaced by new ships of the most up-to-date configuration. This assumption leads to the following change in fleet characteristics over the projection period:

- New ships have engines rated at the current SFC, so even though there are no further improvements in specific fuel consumption, the fuel efficiency of the fleet as a whole will improve over time through retirement and replacement.
- New ships will weigh as much as the average ship built in 2005, so the total cargo capacity of the fleet will increase over time as smaller ships retire and are replaced.
- Container ships will increase in size over time on the trade routes between Asia to either North America or Europe.

2.4.2.4 Trade Analysis by Commodity Type and Trade Route

Determining the total number of days at sea and in port requires information on the relative amount of each commodity that is carried by the different ship type size categories on each of the trade routes. For example, to serve the large crude oil trade from the Middle East Gulf region to the

Gulf Coast of the U.S., 98 percent of the deadweight tonnage is carried on very large oil tankers, while the remaining 2 percent is carried on smaller Suezmax vessels. After the vessel type size distribution was found, voyage parameters were estimated. Specifically, these are days at sea and in port for each voyage (based on ports called, distance between ports, and ship speed**)**, and the number of voyages (based on cargo volume projected by GI and the DTW from Clarkson's Shipping Database). The length of each voyage and number of voyages were used to estimate the total number of days at sea and at port, which is a parameter used later to calculate total fuel consumption for each vessel type and size category over each route and for each commodity type. (More information on determining the round trip distance for each voyage that is associated with cargo demand for the U.S. is provided in Section 2.4.2.5.)

The days at sea were calculated by dividing the round trip distance by the average vessel speed:

Equation 2-7

$$\text{Days at Sea Per Voyage}_{v,s,route} = \frac{\text{round trip distance route}}{\text{speed}_{v,s} \times 24\text{ hrs}}$$

Where:

v = Vessel type
s = Vessel size category
route = Unique trip itinerary
round trip route distance = Trip length in nautical miles
speed = Vessel speed in knots or nautical miles per hour
24 hrs = Number of hours in one day

Table 2-19 presents the speeds by vessel type that were used in the analysis.[34] These values are the same for all size categories, and are assumed to remain constant over the forecast period.

Table 2-19 Vessel Speed by Type

Vessel Type	Speed (knots)
Crude Oil Tankers	13.2
Petroleum Product Tankers	13.2
Chemical Tankers	13.2
Natural Gas Carriers	13.2
Dry Bulk Carriers	14.1
General Cargo Vessels	12.3
Container Vessels	19.9
Other	12.7

The number of voyages along each route for each trade was estimated for each vessel type *v* and size category *s* serving a given route by dividing the tons of cargo moved by the amount of cargo (DTW) per voyage:

Equation 2-8

$$\text{Number of Voyages}_{v,s,trade} = \frac{\text{total metric tonnes of cargo moved}}{\text{fleet average DWT}_{v,s} \times \text{utilization rate}}$$

Where:

v = Vessel type
s = Vessel size category
trade = Commodity type
Fleet average DWT = Median dead weight tonnage carrying capacity in metric tons
Utilization rate = Fraction of total ship DWT capacity used

The cargo per voyage is based on the fleet average ship size from the vessel profile analysis. For most cargo, a utilization rate of 0.9 is assumed to be constant throughout the forecast period. Lowering this factor would increase the estimated number of voyages required to move the forecasted cargo volumes, which would lead to an increase in estimated fuel demand.

In addition to calculating the average days at sea per voyage, the average days in port per voyage was also estimated by assuming that most types of cargo vessels spend four days in port per voyage. RTI notes, however, that this can vary somewhat by commodity and port.

2.4.2.5 Worldwide Estimates of Fuel Demand

This section describes how the information from the vessel and trade analyses were used to calculate the total annual fuel demand associated with international cargo trade. Specifically, for each year *y* of the analysis, the total bunker fuel demand is the sum of the fuel consumed on each route of each trade (commodity). The fuel consumed on each route of each trade is in turn the sum of the fuel consumed for each route and trade for that year by propulsion main engines and auxiliary engines when operated at sea and in port. These steps are illustrated by the following equations:

Equation 2-9

$$\begin{aligned} FC_y &= \sum_{trade} \sum_{route} FC_{trade,\,route,\,year} \\ &= \sum_{trade} \sum_{route} \left[AFC_{trade,\,route,\,y\,at\,sea} \text{ x Days at Sea}_{trade,\,route,\,y} + AFC_{trade,\,route,\,y\,at\,port} \text{ x Days at Port}_{trade,\,route,\,y} \right] \end{aligned}$$

Where:

FC = Fuel consumed in metric tonnes
y = calendar year
trade = Commodity type
route = Unique trip itinerary
AFC = Average daily fuel consumption in metric tonnes
yatsea = Calendar year main and auxiliary engines are operated at sea
yatport = Calendar year main and auxiliary engines are operated in port

Equations 2-10

$$\text{AFC}_{\text{trade, route, y at sea}} = \sum_{\text{v,s,t,r}} (\text{Percent of trade along route})_{\text{v,s}} \left[\text{Fleet AFC}_{\text{v,s}} \text{ x} \left(\text{MELF} + \text{AE at sea LF}\right)\right]$$

$$\text{AFC}_{\text{trade, route, y at port}} = \sum_{\text{v,s,t,r}} (\text{Percent of trade along route})_{\text{v,s}} \left[\text{Fleet AFC}_{\text{v,s}} \text{ x AE import LF}\right]$$

$$\text{Days at Sea}_{\text{trade, route, y}} = \sum_{\text{v,s,t,r}} (\text{Percent of trade along route})_{\text{v,s}} \left[\text{Days at sea per voyage}_{\text{v,s}} \text{ x Number of voyages}_{\text{v,s}}\right]$$

$$\text{Days at Port}_{\text{trade, route, y}} = \sum_{\text{v,s,t,r}} (\text{Percent of trade along route})_{\text{v,s}} \left[\text{Days at port per voyage x Number of voyages}\right]$$

Where:

- AFC = Average daily fuel consumption in metric tones
- *trade* = Commodity type
- *route* = Unique trip itinerary
- *yatsea* = Calendar year main and auxiliary engines are operated at sea
- *yatport* = Calendar year main and auxiliary engines are operated in port
- *y* = calendar year
- *v* = Vessel type
- *s* = Vessel size category
- *t* = Trade
- *r* = Route
- Fleet AFC = Average daily fuel consumption in metric tonnes at full engine load
- MELF = main engine load factor, unitless
- AE at sea LF = auxiliary engine at-sea load factor, unitless
- AE in port LF = auxiliary engine in-port load factor, unitless

The inputs for these last four equations are all derived from the vessel analysis in Section 2.4.2.3 and the trade analysis in Section 2.4.2.2.

2.4.2.6 Worldwide Bunker Fuel Consumption

Based on the methodology outlined above, estimates of global fuel consumption over time were computed, and growth rates determined from these projections.

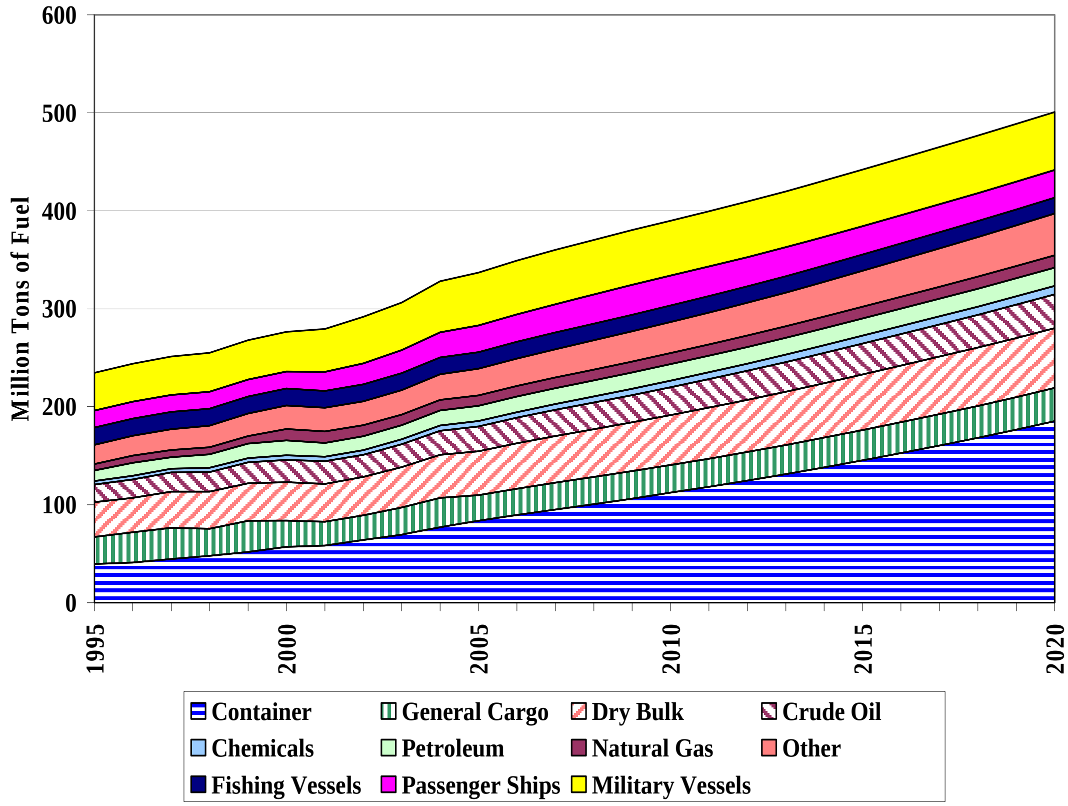


Figure 2-6 Worldwide Bunker Fuel Consumption

Figure 2-6 shows estimated world-wide bunker fuel consumption by vessel type. Figure 2-7 shows the annual growth rates by vessel-type/cargo that are used in the projections shown in Figure 2-6. Total annual growth is generally between 2.5 percent and 3.5 percent over the time period between 2006 and 2020 and generally declines over time, resulting in an average annual growth of around 2.6 percent.

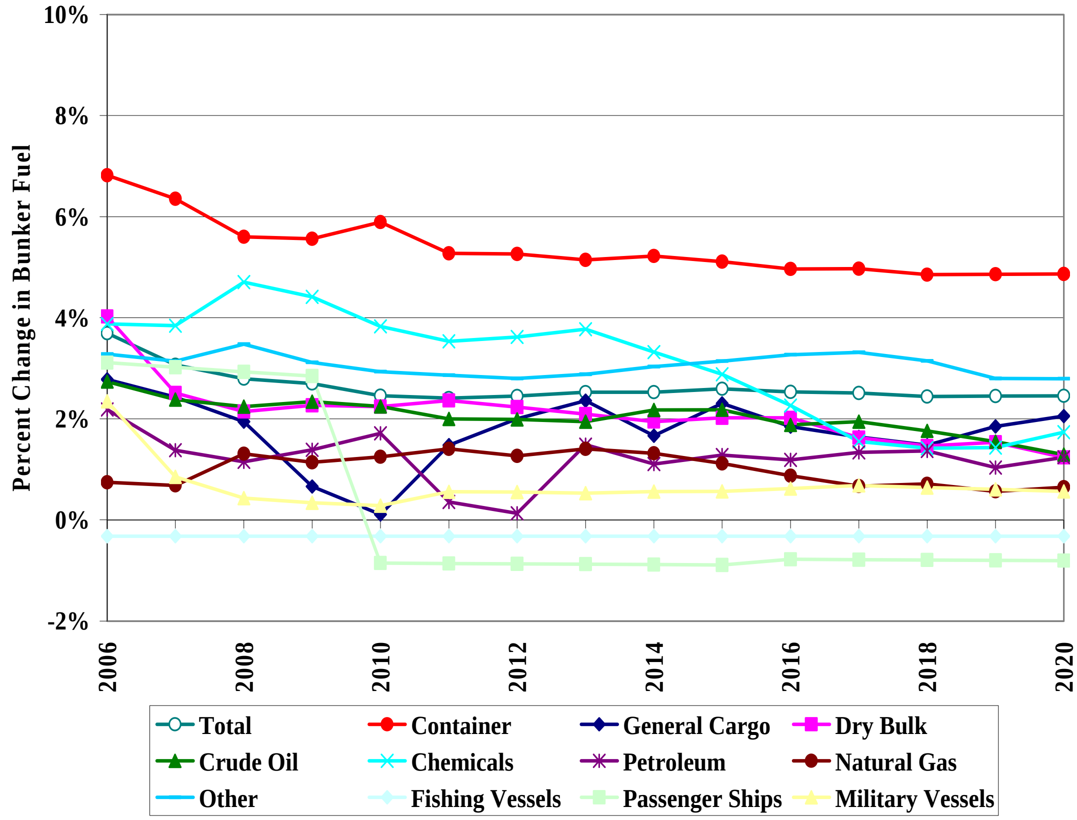


Figure 2-7 Annual Growth Rate in World-Wide Bunker Fuel Use by Commodity Type

2.4.2.7 Fuel Demand Used to Import and Export Cargo for the United States

The methodology described above provides an estimate of fuel consumption for international cargo worldwide. RTI also estimated the subset of fuel demand for cargo imported to and exported from five regions of the U.S. The five regions are:

- North Pacific
- South Pacific
- Gulf
- East Coast
- Great Lakes

For this analysis, the same equations were used, but were limited to routes that carried cargo between specific cities in Asia, Europe and Middle East to the various ports in the specific regions of the U.S.

The trip distances for non-container vessel types were developed from information from Worldscale Association and Maritime Chain.[35,36] The data from Worldscale is considered to be the industry standard for measuring port-to-port distances, particularly for tanker traffic. The reported distances account for common routes through channels, canals, or straits. This distance information was supplemented by data from Maritime Chain, a web service that provides port-to-port distances along with some information about which channels, canals, or straits must be passed on the voyage.

Voyage distances for container vessels are based on information from Containerization International Yearbook (CIY)[37] and calculations by RTI. That reference provides voyage information for all major container services. Based on the frequency of the service, number of vessels assigned to that service, and the number of days in operation per year, RTI estimated the average length of voyages for the particular bilateral trade routes in the Global Insights trade forecasts.

The distance information developed above was combined with the vessel speeds previously shown in Table 2-19 to find the length of a voyage in days. Table 2-20 presents the day lengths for non-containerized vessel types and Table 2-21 shows the same information for container vessels.

Table 2-20 Day Length for Voyages for Non-Container Cargo Ship (approximate average)

	Days per Voyage				
Global Insights Trade Regions	US South Pacific	US North Pacific	US East Coast	US Great Lakes	US Gulf
Africa East-South	68	75	57	62	54
Africa North-Mediterranean	49	56	37	43	47
Africa West	56	63	36	46	43
Australia-New Zealand	48	47	65	81	63
Canada East	37	46	7	18	19
Canada West	11	5	40	58	39
Caspian Region	95	89	41	46	48
China	41	36	73	87	69
Europe Eastern	61	68	38	45	46
Europe Western-North	53	60	24	32	34
Europe Western-South	54	61	30	37	37
Greater Caribbean	26	33	16	29	17
Japan	35	31	65	81	62
Middle East Gulf	77	72	56	65	83
Pacific High Growth	52	48	67	76	88
Rest of Asia	68	64	66	64	73
Russia-FSU	64	71	38	46	48
Rest of South America	51	30	41	46	44

Table 2-21 Day Length for Voyages for Container-Ship Trade Routes

Origin – Destination Regions	**Days per Voyage**
Asia – North America (Pacific)	37
Europe – North America (Atlantic)	37
Mediterranean – North America	41
Australia/New Zealand – North America	61
South America – North America	48
Africa South – North America (Atlantic)	54
Africa West – North America (Atlantic)	43

Origin – Destination Regions	Days per Voyage
Asia – North America (Atlantic)	68
Europe – North America (Pacific)	64
Africa South – North America (Pacific)	68
Africa West – North America (Pacific)	38
Caspian Region – North America (Atlantic)	42
Caspian Region – North America (Pacific)	38
Middle East/Gulf Region – North America (Atlantic)	63
Middle East/Gulf Region – North America (Pacific)	80

2.4.2.8 Bunker Fuel Consumption for the United States

Figure 2-8 and Figure 2-9 present the estimates of fuel use for delivering trade goods to and from the U.S. The results in Figure 2-8 show estimated historical bunker fuel use in year 2001 of around 47 million tonnes (note: while this fuel is used to carry trade goods to and from the U.S., it is not necessarily all purchased in the U.S. and is not all burned in U.S. waters). This amount grows to over 90 million tonnes by 2020 with the most growth occurring on trade routes from the East Coast and the "South Pacific" region of the West Coast.

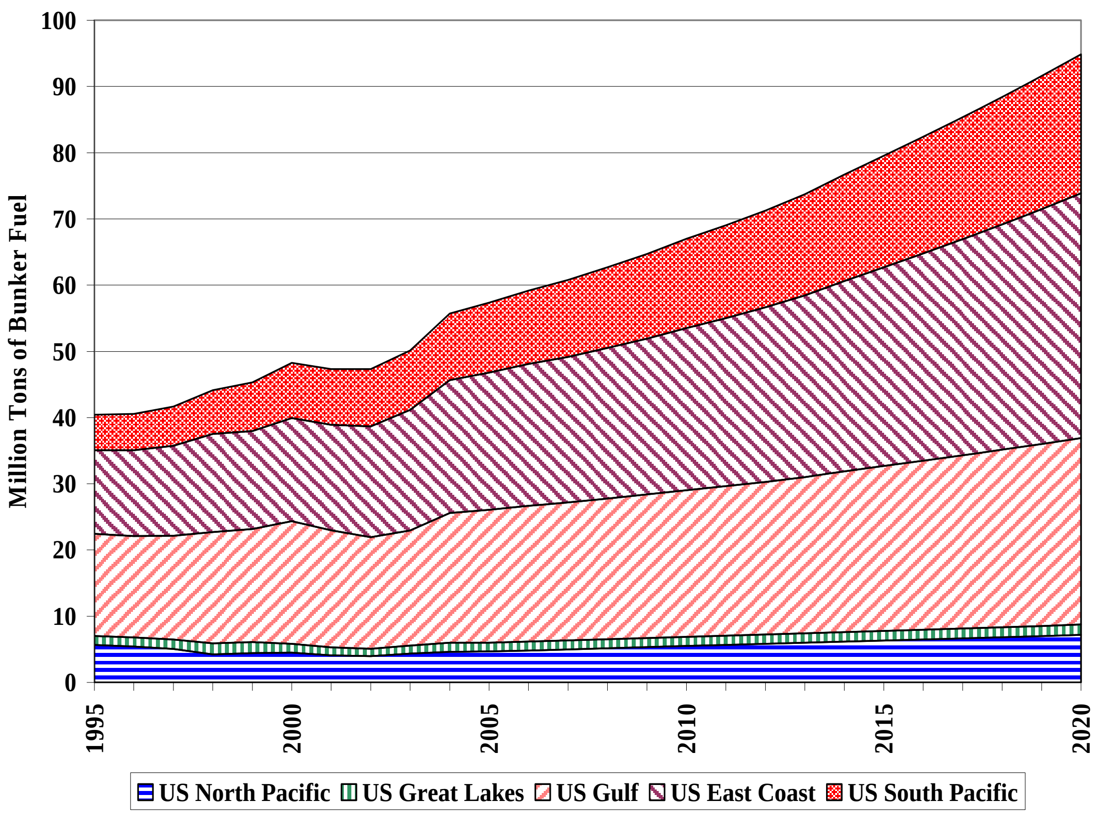


Figure 2-8 Bunker Fuel Used to Import and Export Cargo by Region of the United States

Figure 2-9 shows the estimated annual growth rates for the fuel consumption that are used in the projections shown in Figure 2-8. Overall, the average annual growth rate in marine bunkers associated with future U.S. trade flows is 3.4 percent between 2005 and 2020.

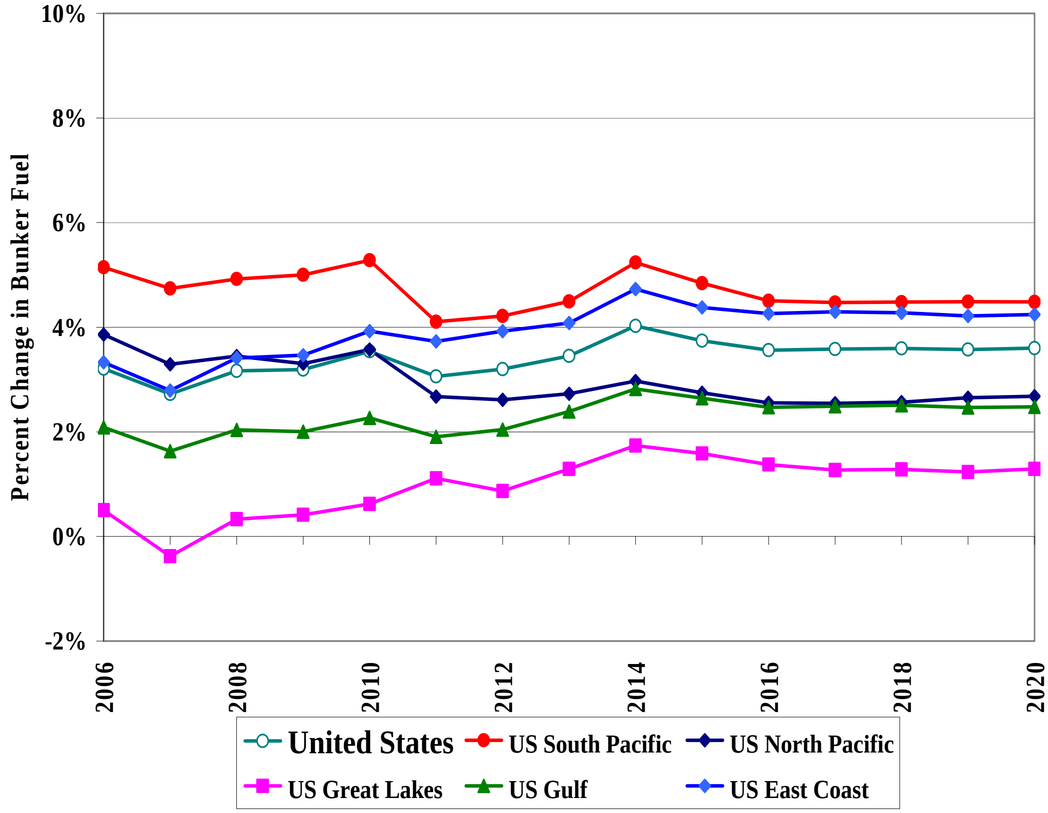


Figure 2-9 Annual Growth Rates for Bunker Fuel Used to Import and Export Cargo by Region of the United States

2.4.2.9 2020 Growth Factors for Nine Geographic Regions

The results of the RTI analysis described above are used to develop the growth factors that are necessary to project the 2002 base year emissions inventory to 2020. The next two sections describe how the five RTI regions were associated with the nine regions analyzed in this report, and how the specific growth rates for each of the nine regions were developed.

2.4.2.9.1 Mapping the RTI Regional Results to the Nine Region Analysis

The nine geographic regions analyzed in this study were designed to be consistent with the five RTI regional modeling domains. More specifically, four of the nine geographic areas in this study, i.e., Alaska East, Alaska West, Hawaii East, and Hawaii West are actually subsets of two broader regional areas that were analyzed by RTI, i.e., the North Pacific for both Alaska regions and South Pacific for Hawaii. Therefore, the growth rate information from the related larger region was assumed to be representative for that state.

The nine geographic regions represented in the emission inventory study are presented in Figure 2-1. The association of the RTI regions to the emission inventory regions is shown in Table 2-22.

Table 2-22 Association of the RTI Regions to the Nine Emission Inventory Regions

Consumption Region	Corresponding Emission Inventory Region
North Pacific	North Pacific (NP)
North Pacific	Alaska East (AE)
North Pacific	Alaska West (AW)
South Pacific	South Pacific (SP)
South Pacific	Hawaii East (HE)
South Pacific	Hawaii West (HW)
Gulf	Gulf Coast (GC)
East Coast	East Coast (EC)
Great Lakes	Great Lakes (GL)

2.4.2.9.2 Growth Factors for the Emission Inventory Analysis

Emission inventories for 2020 are estimated by multiplying the 2002 baseline inventory for each region by a corresponding growth factor that was developed from the RTI regional results. Specifically, the average annual growth rate from 2002-2020 was calculated for each of the five regions. Each regional growth rate was then compounded over the inventory projection time period for 2020, i.e., 18 years. The resulting multiplicative growth factors for each emission inventory region and the associated RTI average annual growth rates are presented in Table 2-23 for 2020.

Table 2-23 Regional Emission Inventory Growth Factors for 2020

Emission Inventory Region	2002-2020 Average Annualized Growth Rate (%)	Multiplicative Growth Factor Relative to 2002
Alaska East (AE)	3.3	1.79
Alaska West (AW)	3.3	1.79
East Coast (EC)	4.5	2.21
Gulf Coast (GC)	2.9	1.67
Hawaii East (HE)	5.0	2.41
Hawaii West (HW)	5.0	2.41
North Pacific (NP)	3.3	1.79
South Pacific (SP)	5.0	2.41
Great Lakes (GL)	1.7	1.35

2.4.3 Emission Controls in 2020 Baseline and Control Scenarios

This section describes the control programs present in the 2020 baseline and control scenarios. Section 2.4.4 describes the process of incorporating these programs into the 2020 emission inventories.

The baseline scenario includes the International Marine Organization's (IMO) Tier I NO_X standard for marine diesel engines that became effective in 2000, as well as the Tier II standard that will become effective in 2011. Also included in the baseline inventories is the NO_X retrofit program for pre-controlled engines proposed by IMO.

Although the 0.1% fuel sulfur requirement goes into place for all vessels operating in ECAs beginning in 2015, the use of 2020 as the analytic year will still provide a representative scenario for the impact of the 0.1% fuel sulfur requirement on human health and the environment. This is because the fuel requirements of the ECA go into effect all at once; there is no phase-in. So the impacts of the fuel requirement in 2020 are expected to be the same as in 2015, with a small increase due to growth. With regard to the NO_X impacts, while 2020 will include five years of turnover to the Tier III standards, the long service lives of engines on ocean-going vessels mean that these impacts will be small and affect less than 25% of the total fleet, assuming an average 20-year service life. These NO_X reductions would not inflate the benefits of the program by very much, if any. Note that the global fuel sulfur standard does not go into effect until 2020. We did not include this in the 2020 analysis, to provide a better estimate of benefits in the early (pre-2020) years of the program

The effects of these controls are reflected in the 2020 emission inventories by applying appropriate adjustment factors that reflect the percentage of the vessel fleet in those years that are estimated to comply with the controls. Adjustment factors are ratios of 2020 to 2002 calendar year (CY) emission factors (EFs). Adjustment factors are derived separately by engine type for propulsion and auxiliary engines. The adjustment factors for propulsion engines are applied to the propulsion portion of the port inventory and the interport portion of the inventory. The adjustment factors for auxiliary engines are applied to the auxiliary portion of the port inventory.

The control scenario includes an Emission Control Area (ECA) within a distance of 200 nautical miles (nm) from shore. Outside this distance, baseline controls were applied (i.e., the Tier I and Tier II NO_X standards, the NO_X retrofit program, and current fuel sulfur content levels). The ECA NO_X controls include the baseline controls above, plus Tier II NO_X standards. Fuel sulfur content is also assumed to be controlled to 1,000 ppm within the ECA. Note that gas and steam turbine engines are not subject to any of the NO_x standards; however, these engines are not a large part of the inventory.

The retrofit program for Tier 0 (pre-control) engines was modeled as 11 percent control from Tier 0 for 80 percent of 1990 thru 1999 model year (MY) engines greater than 90 liters per cylinder (L/cyl) starting in 2011. The retrofit program was also modeled with a five year phase-in. The current Tier I controls, which also are modeled as achieving an 11 percent reduction from Tier 0, apply to the 2000 thru 2010 MY engines. In 2011 thru 2015, Tier II controls are applied. Tier II controls are modeled as a 2.5 g/kW-hr reduction from Tier I. In the ECA area only, for 2016 MY engines and beyond, Tier III controls are applied. Tier III controls are modeled as achieving an 80 percent reduction from Tier I levels. Control of fuel sulfur content within the ECA area affects SO_2 and PM emissions.

ECA controls were applied to the 48 state region as well as Alaska East (AE) and Hawaii East (HE). Alaska West (AW) and Hawaii West (HW) are baseline cases only.

2.4.4 2020 Emission Factors

This section describes the emission factors that are used in the 2020 scenarios. HC and CO emission factors are assumed to remain unchanged from the 2002 scenario. NO_x and fuel sulfur controls are anticipated to lower NO_X, SO_2 and PM emission factors. The switch to lower sulfur distillate fuel use is also assumed to lower CO_2 emissions slightly.

The NO_X emission factors (EFs) by engine/ship type and tier are provided in Table 2-24. Tier 0 refers to pre-control. There are separate entries for Tier 0/1/2 base and Tier 0/1/2 control, since the control engines would be using distillate fuel, and there are small NO_X emission reductions assumed when switching from residual to distillate fuel.[17] The NO_X control EFs by tier were derived using the assumptions described in section 2.4.3.

Table 2-24 Modeled NO_X Emission Factors by Tier

Engine/ Ship Type	NO_X EF (g/kW-hr)								
	Baseline				Control Areas				
	Tier 0	T0 retrofit	Tier I	Tier II	Tier 0	T0 retrofit	Tier I	Tier II	Tier III
Main									
SSD	18.1	16.1	16.1	13.6	17	15.1	15.1	12.6	3
MSD	14	12.5	12.5	10.0	13.2	11.7	11.7	9.2	2.3
ST	2.1	n/a	n/a	n/a	2	n/a	n/a	n/a	n/a
GT	6.1	n/a	n/a	n/a	5.7	n/a	n/a	n/a	n/a
Aux									
Pass	14.6	n/a[a]	13.0	10.5	14.6	n/a[a]	13.0	10.5	2.6
Other	14.5	n/a[a]	12.9	10.4	14.5	n/a[a]	12.9	10.4	2.6

[a] The retrofit program applies to engines over 90 L/cyl; auxiliary engines are smaller than this cutpoint and would therefore not be subject to the program.

The NO_X EFs by tier were then used with the vessel age distributions (Table 2-25 & Table 2-26) to generate calendar year NO_X EFs by engine/ship type for the base and control areas included in the scenarios. These calendar year NO_X EFs are provided in Table 2-27 below. Since the age distributions are different for vessels in the Great Lakes, NO_X EFs were determined separately for the Great Lakes.

Table 2-25 Vessel Age Distribution for Deep Sea Ports by Engine Type

Age Group (years old)	Propulsion Engine Type [a] (Fraction of Total)				All Auxiliary Engines
	MSD	SSD	GT	ST	
0	0.00570	0.02667	0.00000	0.00447	0.01958
1	0.07693	0.07741	0.07189	0.12194	0.07670
2	0.10202	0.07512	0.14045	0.16464	0.08426
3	0.08456	0.07195	0.05608	0.05321	0.07489
4	0.08590	0.05504	0.67963	0.00000	0.07831
5	0.06427	0.05563	0.04165	0.00000	0.05685
6	0.06024	0.04042	0.00000	0.00000	0.04455
7	0.07867	0.07266	0.00626	0.00000	0.07150
8	0.06730	0.05763	0.00000	0.00000	0.05764
9	0.04181	0.04871	0.00000	0.00000	0.04475
10	0.04106	0.04777	0.00000	0.00000	0.04364
11	0.03100	0.03828	0.00000	0.00000	0.03538
12	0.04527	0.03888	0.00000	0.04873	0.04160
13	0.03583	0.02787	0.00000	0.00000	0.02909
14	0.03519	0.02824	0.00000	0.00000	0.02935
15	0.02921	0.01466	0.00000	0.00000	0.01869
16	0.00089	0.01660	0.00000	0.00000	0.01189
17	0.01326	0.01582	0.00000	0.00000	0.01462
18	0.00847	0.02414	0.00000	0.00000	0.01966
19	0.00805	0.01982	0.00000	0.00000	0.01550
20	0.00566	0.02258	0.00000	0.00000	0.01756
21	0.00495	0.02945	0.00000	0.00000	0.02260
22	0.00503	0.01883	0.00000	0.00875	0.01467
23	0.00676	0.01080	0.00000	0.00883	0.00943
24	0.00539	0.01091	0.00000	0.00883	0.00900
25	0.01175	0.01099	0.00000	0.18029	0.01224
26	0.00803	0.01045	0.00000	0.11065	0.01130
27	0.00522	0.00835	0.00000	0.01395	0.00738
28	0.00294	0.00788	0.00000	0.08657	0.00659
29	0.00285	0.00370	0.00034	0.02907	0.00349
30	0.00254	0.00106	0.00370	0.05126	0.00193
31	0.00084	0.00113	0.00000	0.00605	0.00096
32	0.00023	0.00367	0.00000	0.07105	0.00322
33	0.00117	0.00582	0.00000	0.00000	0.00419
34	0.00132	0.00092	0.00000	0.00000	0.00098
35+	0.01967	0.00013	0.00000	0.03172	0.00598

[a] MSD is medium speed diesel, SSD is slow speed diesel, GT is gas turbine, ST is steam turbine.

Table 2-26 Vessel Age Distribution for Great Lake Ports by Engine Type

Age Group (years old)	Propulsion Engine Type[a] (Fraction of Total)			
	MSD	SSD	ST	All Auxiliary Engines
0	0.01610	0.03913	0.00000	0.02399
1	0.02097	0.03489	0.00000	0.02243
2	0.01370	0.04644	0.00000	0.02544
3	0.02695	0.03040	0.00000	0.02511
4	0.01571	0.04547	0.00000	0.02497
5	0.04584	0.01498	0.00000	0.02442
6	0.01494	0.02180	0.00000	0.01528
7	0.01327	0.01857	0.00000	0.01391
8	0.00099	0.04842	0.00000	0.02107
9	0.00027	0.03376	0.00000	0.01454
10	0.01085	0.01177	0.00000	0.01076
11	0.00553	0.01183	0.00000	0.00782
12	0.00739	0.00546	0.00000	0.00626
13	0.02289	0.02557	0.00000	0.02242
14	0.00000	0.00286	0.00000	0.00121
15	0.00275	0.00510	0.00000	0.00361
16	0.00069	0.00073	0.00000	0.00078
17	0.00000	0.00104	0.00000	0.00041
18	0.00342	0.01967	0.00000	0.01059
19	0.00219	0.01220	0.00000	0.00645
20	0.00867	0.06140	0.00000	0.03034
21	0.00000	0.05638	0.00000	0.02503
22	0.03375	0.02108	0.00000	0.02279
23	0.04270	0.02051	0.00000	0.02606
24	0.08161	0.01010	0.00000	0.03744
25	0.02935	0.05217	0.00000	0.03480
26	0.18511	0.00522	0.00000	0.07701
27	0.01870	0.00389	0.00000	0.01083
28	0.13815	0.01438	0.00000	0.06181
29	0.05487	0.01160	0.00000	0.02697
30	0.00000	0.00114	0.00000	0.00047
31	0.03986	0.00000	0.00000	0.01611
32	0.03654	0.00282	0.00000	0.01631
33	0.03358	0.00000	0.00000	0.01358
34	0.00295	0.00123	0.00000	0.00165
35+	0.06974	0.30796	1.00000	0.31734

[a] MSD is medium speed diesel, SSD is slow speed diesel, GT is gas turbine, ST is steam turbine.

[b] Fleet average weighted by installed power (ship port calls x main propulsion engine power).

Table 2-27 Modeled NO_X Emission Factors by Calendar Year and Control Type

Engine/ Ship Type	CY NO_X EF (g/kW-hr)				
		2020 Base		2020 ECA Control	
	2002	DSP	GL	DSP	GL
Main					
SSD	18.1	14.7	15.9	10.8	13.1
MSD	14	10.9	13.1	7.7	11.8
ST	2.1	2.1	2.1	2.0	2.0
GT	6.1	6.1	n/a	5.7	n/a
Aux					
Pass	14.6	11.7	13.6	8.6	12.0
Other	14.5	11.5	13.4	8.6	12.0

DSP = Deep water ports and areas other than the Great Lakes
GL = Great Lakes

The PM and SO_2 EFs are a function of fuel sulfur level. For the baseline portions of the inventory, there are two residual fuel sulfur levels modeled: 25,000 ppm for the West Coast and 27,000 ppm for the rest of the U.S. The baseline distillate fuel sulfur level assumed for all areas is 15,000 ppm. As discussed in section 2.3.2.3.5, for the baseline, main engines use residual fuel and auxiliary engines use a mix of residual and distillate fuel. For the control areas, there is one level of distillate fuel sulfur assumed to be used by all engines: 1,000 ppm for the ECA control areas.

Table 2-28 provides the PM_{10} EFs by engine/ship type and fuel sulfur level. For modeling purposes, $PM_{2.5}$ is assumed to be 92 percent of PM_{10}. The PM EFs are adjusted to reflect the appropriate fuel sulfur levels using Equation 2-2.

Table 2-29 provides the modeled SO_2 EFs. SO_2 emission reductions are directly proportional to reductions in fuel sulfur content.

CO_2 is directly proportional to fuel consumed. Table 2-30 provides the modeled CO_2 and brake specific fuel consumption (BSFC) EFs. Due to the higher energy content of distillate fuel on a mass basis, the switch to distillate fuel for the control areas results in a small reduction to BSFC and, correspondingly, CO_2 emissions.[17]

Table 2-28 Modeled PM_{10} Emission Factors

Engine/ Ship Type	PM_{10} EF (g/kW-hr)		
	Baseline		Control Areas
	Other than West Coast 27,000 ppm S	West Coast[a] 25,000 ppm S	ECA 1,000 ppm S
Main			
SSD	1.40	1.40	0.19
MSD	1.40	1.40	0.19
ST	1.50	1.40	0.17
GT	1.50	1.40	0.17
Aux			
Pass	1.40	1.30	0.19
Other	1.20	1.10	0.19

[a] For the base cases, the West Coast fuel is assumed to be used in the following regions: Alaska East (AE), Alaska West (AW), Hawaii East (HE), Hawaii West (HW), North Pacific (NP), and South Pacific (SP).

Table 2-29 Modeled SO_2 Emission Factors*

Engine/ Ship Type	SO_2 EF (g/kW-hr)		
	Baseline		Control Areas
	Other than West Coast 27,000 ppm S	West Coast[a] 25,000 ppm S	ECA Control 1,000 ppm S
Main			
SSD	10.29	9.53	0.36
MSD	11.09	10.26	0.39
ST	16.10	14.91	0.57
GT	16.10	14.91	0.57
Aux			
Pass	10.70	9.93	0.39
Other	9.66	9.07	0.39

[a] For the base cases, the West Coast fuel is assumed to be used in the following regions: Alaska East (AE), Alaska West (AW), Hawaii East (HE), Hawaii West (HW), North Pacific (NP), and South Pacific (SP).

Table 2-30 Modeled Fuel Consumption and CO_2 Emission Factors

Engine/ Ship Type	EF (g/kW-hr)			
	Baseline		Control Areas	
	BSFC	CO_2	BSFC	CO_2
Main				
SSD	195	620	185	589
MSD	210	668	200	637
ST	305	970	290	923
GT	305	970	290	923
Aux				
Pass	210	668	200	636
Other	210	668	200	636

2.4.5 Calculation of 2020 Near Port and Interport Inventories

Based on the emission factors described in Section 2.4.4, appropriate adjustments were applied to the NO_X, PM (PM_{10} and $PM_{2.5}$), SO_2, and CO_2 inventory of each 2020 scenario. This section describes the development and application of the adjustment factors to the port and interport inventories, and the methodology for combining the port and interport portions.

2.4.5.1 Port Methodology

2.4.5.1.1 Non-California Ports

For the non-California ports, 2002 emissions for each port are summed by engine/ship type. Propulsion and auxiliary emissions are summed separately, since the EF adjustment factors differ. The appropriate regional growth factor, as provided in Table 2-23, is then applied, along with EF adjustment factors by engine/ship type. The EF adjustment factors are a ratio of the control EF to the 2002 EF. Table 2-31 through Table 2-35 provide the EF adjustment factors for each pollutant and control area. The ports will be subject to ECA controls in the control scenario. These tables are also used as input for the California ports and interport control inventory development, discussed in subsequent sections.

Table 2-31 NO_X EF Adjustment Factors by Engine/Ship Type and Control Type[a]

Engine/ Ship Type	2020 Base		2020 ECA Control	
	DSP	GL	DSP	GL
Main				
SSD	0.8130	0.8783	0.5967	0.7219
MSD	0.7804	0.9366	0.5515	0.8423
ST	1.0000	1.0000	0.9524	0.9524
GT	1.0000	n/a	0.9344	n/a
Aux				
Pass	0.7985	0.9296	0.5869	0.8196
Other	0.7972	0.9292	0.5940	0.8295

[a] NO_X adjustment factors are a ratio of future base or control EFs to 2002 EFs
DSP = deep water ports and areas other than the Great Lakes; GL = Great Lakes

Table 2-32 PM_{10} EF Adjustment Factors by Engine/Ship Type and Control Type[a]

Engine/ Ship Type	2020 Base		2020 ECA Control	
	Other	WC	Other	WC
Main				
SSD	1.0000	1.0000	0.1352	0.1352
MSD	1.0000	1.0000	0.1328	0.1328
ST	1.0000	1.0000	0.1108	0.1187
GT	1.0000	1.0000	0.1108	0.1187
Aux				
Pass	1.0000	1.0000	0.1328	0.1430
Other	1.0000	1.0000	0.1550	0.1691

[a] PM_{10} adjustment factors are a ratio of the control EFs to the 2002 EFs. PM is not adjusted for the future baseline because fuel sulfur levels are only assumed to change within the ECA.
Other = Other than West Coast
WC = Ports/areas within the West Coast. This includes the regions of Alaska, Hawaii, North Pacific, and South Pacific.

Table 2-33 $PM_{2.5}$ EF Adjustment Factors by Engine/Ship Type and Control Type[a]

Engine/ Ship Type	2020 Base		2020 ECA Control	
	Other	WC	Other	WC
Main				
SSD	1.0000	1.0000	0.1339	0.1339
MSD	1.0000	1.0000	0.1316	0.1316
ST	1.0000	1.0000	0.1092	0.1176
GT	1.0000	1.0000	0.1092	0.1176
Aux				
Pass	1.0000	1.0000	0.1316	0.1426
Other	1.0000	1.0000	0.1555	0.1711

[a] $PM_{2.5}$ adjustment factors are a ratio of the control EFs to the 2002 EFs. PM is not adjusted for the future baseline because fuel sulfur levels are only assumed to change within the ECA. The $PM_{2.5}$ adjustment factors are slightly different from those for PM_{10} due to rounding.
Other = Other than West Coast
WC = Ports/areas within the West Coast. This includes the regions of Alaska, Hawaii, North Pacific, and South Pacific.

Table 2-34 SO_2 EF Adjustment Factors by Engine/Ship Type and Control Type[a]

Engine/ Ship Type	2020 Base		2020 ECA Control	
	Other	WC	Other	WC
Main				
SSD	1.0000	1.0000	0.0351	0.0380
MSD	1.0000	1.0000	0.0353	0.0381
ST	1.0000	1.0000	0.0352	0.0380
GT	1.0000	1.0000	0.0352	0.0380
Aux				
Pass	1.0000	1.0000	0.0365	0.0394
Other	1.0000	1.0000	0.0405	0.0431

[a] SO_2 adjustment factors are a ratio of the control EFs to the 2002 EFs. SO_2 is not adjusted for the future baseline because fuel sulfur levels are only assumed to change within the ECA.
Other = Other than West Coast
WC = Ports/areas within the West Coast. This includes the regions of Alaska, Hawaii, North Pacific, and South Pacific.

Table 2-35 CO_2 EF Adjustment Factors by Engine/Ship Type and Control Type[a]

Engine/ Ship Type	2020 Base		2020 ECA Control	
	Other	WC	Other	WC
Main				
SSD	1.0000	1.0000	0.9488	0.9488
MSD	1.0000	1.0000	0.9531	0.9531
ST	1.0000	1.0000	0.9509	0.9509
GT	1.0000	1.0000	0.9509	0.9509
Aux				
Pass	1.0000	1.0000	0.9525	0.9593
Other	1.0000	1.0000	0.9525	0.9683

[a] CO_2 adjustment factors are a ratio of the control EFs to the 2002 EFs. CO_2 is not adjusted for the future baseline because fuel consumption (BSFC) is only assumed to change within the ECA.
Other = Other than West Coast
WC = Ports/areas within the West Coast. This includes the regions of Alaska, Hawaii, North Pacific, and South Pacific.

2.4.5.1.2 California Ports

For the California ports, 2002 emissions for each port are summed by ship type. Propulsion and auxiliary emissions are summed separately, since the EF adjustment factors differ. The EF adjustment factors by engine/ship type, provided in the previous section, are consolidated by ship type, using the CARB assumption that engines on all ships except passenger ships are 95 percent slow speed diesel (SSD) engines and 5 percent medium speed diesel engines (MSD) based upon a 2005 ARB survey.[C] All passenger ships were assumed to be medium speed diesel engines with electric drive propulsion (MSD-ED). Steam turbines (ST) and gas-turbines (GT) are not included in

[C] California Air Resources Board, *2005 Oceangoing Ship Survey, Summary of Results*, September 2005.

the CARB inventory. The EF adjustment factors by ship type are then applied, along with ship-specific growth factors supplied by CARB. The ship-specific growth factors relative to 2002 are provided in Table 2-36 below.

Table 2-36 Growth Factors by Ship Type for California Ports Relative to 2002

Ship Type	Calendar Year	
	2002	2020
Auto	1.0000	1.5010
Bulk	1.0000	0.2918
Container	1.0000	2.5861
General	1.0000	0.7331
Passenger	1.0000	7.5764
Reefer	1.0000	1.0339
RoRo	1.0000	1.5010
Tanker	1.0000	2.0979

2.4.5.2 Interport Methodology

The interport portion of the inventory is not segregated by engine or ship type. As a result, regional EF adjustment factors were developed based on the assumed mix of main (propulsion) engine types in each region. The mix of main engine types by region was developed using the ship call data and is presented in Table 2-37. Main engines are considered a good surrogate for interport emissions, since the majority of emissions while underway are due to the main engines. The EF adjustment factors by main engine type in Section 2.4.5.1were used together with the mix of main engine types by region to develop the EF regional adjustment factors for each control area. The resulting EF regional adjustment factors for each pollutant and control area are provided in Table 2-38 through Table 2-42 below. These EF regional adjustment factors, together with the regional growth factors in Table 2-23, were applied to calculate the future inventories for each control area.

Table 2-37 Installed Power by Main Engine Type and Region

Region	2020 Installed Power (%)				
	MSD	SSD	GT	ST	Total
Alaska East (AE)	19.1%	18.4%	0.3%	62.2%	100%
Alaska West (AW)	19.1%	18.4%	0.3%	62.2%	100%
East Coast (EC)	25.6%	72.5%	0.9%	1.0%	100%
Gulf Coast (GC)	13.7%	85.5%	0.0%	0.8%	100%
Hawaii East (HE)	66.2%	18.5%	7.4%	8.0%	100%
Hawaii West (HW)	66.2%	18.5%	7.4%	8.0%	100%
North Pacific (NP)	5.1%	83.5%	1.6%	9.7%	100%
South Pacific (SP)	29.2%	70.8%	0.0%	0.0%	100%
Great Lakes (GL)	48%	44%	0%	8%	100%

Table 2-38 NO_X EF Adjustment Factors by Region and Control Type[a]

U.S. Region	2002	2020	
		Base	ECA Control
Alaska East (AE)	1.0000	0.9237	0.8104
Alaska West (AW)	1.0000	0.9237	n/a
East Coast (EC)	1.0000	0.8082	0.5917
Gulf Coast (GC)	1.0000	0.8102	0.5935
Hawaii East (HE)	1.0000	0.8202	0.6201
Hawaii West (HW)	1.0000	0.8202	n/a
North Pacific (NP)	1.0000	0.8325	0.6343
South Pacific (SP)	1.0000	0.8036	0.5837
Great Lakes (GL)	1.0000	0.8131	0.7989
Out of Region[b]	1.0000	0.8095	n/a

[a] NO_X adjustment factors are a ratio of future base or control EFs to 2002 EFs. These regional adjustment factors are used to adjust the interport portion of the 2002 inventory.
[b] Out of Region refers to areas outside the 200 nm US modeling boundary, but within the air quality modeling domain. The out of region adjustment factors are derived by weighting the regional adjustment factors by the main propulsion power in each region.

Table 2-39 PM_{10} EF Adjustment Factors by Region and Control Type[a]

U.S. Region	2002	2020	
		Base	ECA Control
Alaska East (AE)	1.0000	1.0000	0.1244
Alaska West (AW)	1.0000	1.0000	n/a
East Coast (EC)	1.0000	1.0000	0.1341
Gulf Coast (GC)	1.0000	1.0000	0.1347
Hawaii East (HE)	1.0000	1.0000	0.1311
Hawaii West (HW)	1.0000	1.0000	n/a
North Pacific (NP)	1.0000	1.0000	0.1332
South Pacific (SP)	1.0000	1.0000	0.1345
Great Lakes (GL)	1.0000	1.0000	0.1320
Out of Region[b]	1.0000	1.0000	n/a

[a] PM_{10} adjustment factors are a ratio of future base or control EFs to 2002 EFs. These regional adjustment factors are used to adjust the interport portion of the 2002 inventory.
[b] Out of Region refers to areas outside the 200 nm US modeling boundary, but within the air quality modeling domain. The out of region adjustment factors are derived by weighting the regional adjustment factors by the main propulsion power in each region.

Table 2-40 $PM_{2.5}$ EF Adjustment Factors by Region and Control Type[a]

U.S. Region	2002	2020	
		Base	ECA Control
Alaska East (AE)	1.0000	1.0000	0.1233
Alaska West (AW)	1.0000	1.0000	n/a
East Coast (EC)	1.0000	1.0000	0.1329
Gulf Coast (GC)	1.0000	1.0000	0.1334
Hawaii East (HE)	1.0000	1.0000	0.1299
Hawaii West (HW)	1.0000	1.0000	n/a
North Pacific (NP)	1.0000	1.0000	0.1320
South Pacific (SP)	1.0000	1.0000	0.1332
Great Lakes (GL)	1.0000	1.0000	0.1307
Out of Regionb	1.0000	1.0000	n/a

[a] $PM_{2.5}$ adjustment factors are a ratio of future base or control EFs to 2002 EFs. These regional adjustment factors are used to adjust the interport portion of the 2002 inventory.
[b] Out of Region refers to areas outside the 200 nm US modeling boundary, but within the air quality modeling domain. The out of region adjustment factors are derived by weighting the regional adjustment factors by the main propulsion power in each region.

Table 2-41 SO_2 EF Adjustment Factors by Region and Control Type[a]

U.S. Region	2002	2020	
		Base	ECA Control
Alaska East (AE)	1.0000	1.0000	0.0380
Alaska West (AW)	1.0000	1.0000	n/a
East Coast (EC)	1.0000	1.0000	0.0352
Gulf Coast (GC)	1.0000	1.0000	0.0352
Hawaii East (HE)	1.0000	1.0000	0.0381
Hawaii West (HW)	1.0000	1.0000	n/a
North Pacific (NP)	1.0000	1.0000	0.0380
South Pacific (SP)	1.0000	1.0000	0.0380
Great Lakes (GL)	1.0000	1.0000	0.0352
Out of Region[b]	1.0000	1.0000	n/a

[a] SO_2 adjustment factors are a ratio of future base or control EFs to 2002 EFs. These regional adjustment factors are used to adjust the interport portion of the 2002 inventory.
[b] Out of Region refers to areas outside the 200 nm US modeling boundary, but within the air quality modeling domain. The out of region adjustment factors are derived by weighting the regional adjustment factors by the main propulsion power in each region are derived by weighting the regional adjustment factors by the main propulsion power in each region.

Table 2-42 CO_2 EF Adjustment Factors by Region and Control Type[a]

U.S. Region	2002	2020	
		Base	ECA Control
Alaska East (AE)	1.0000	1.0000	0.9509
Alaska West (AW)	1.0000	1.0000	n/a
East Coast (EC)	1.0000	1.0000	0.9499
Gulf Coast (GC)	1.0000	1.0000	0.9494
Hawaii East (HE)	1.0000	1.0000	0.9519
Hawaii West (HW)	1.0000	1.0000	n/a
North Pacific (NP)	1.0000	1.0000	0.9493
South Pacific (SP)	1.0000	1.0000	0.9501
Great Lakes (GL)	1.0000	1.0000	0.9510
Out of Region[b]	1.0000	1.0000	n/a

[a] CO_2 adjustment factors are a ratio of future base or control EFs to 2002 EFs. These regional adjustment factors are used to adjust the interport portion of the 2002 inventory**.**

[b] Out of Region refers to areas outside the 200 nm US modeling boundary, but within the air quality modeling domain. The out of region adjustment factors are derived by weighting the regional adjustment factors by the main propulsion power in each region.

2.4.5.3 Estimating and Combining the Near Port and Interport Control Inventories

To produce future year control scenarios, the interport inventories were scaled by a growth factor to 2020, as previously described. An ECA boundary line was drawn so that each point on it was at a 200 nm distance from the nearest point on land. Adjustment factors, as described in section 2.4.4, were then applied to interport emissions within the ECA boundary.

To create control scenarios in the near port inventories, growth and control factors were applied to the 2002 near port inventories (described in sections 2.4.2 and 2.4.4). The near port inventories were then converted into a gridded format (section 2.3.4). Using this grid, STEEM values were removed from near port cells and near port emissions were used as replacement values. In cases where the emissions near ports were only partially attributable to port traffic, the STEEM inventory was reduced rather than removed.

Interport and near port emissions were then aggregated to form regional totals.

2.4.6 2020 Baseline and Control Inventories and Fuel Consumption

The baseline emission inventories for 2020 are presented in Table 2-43.

Table 2-43 2020 Baseline Inventory

U.S. Region	Metric Tonnes per Year						
	NOx	PM10	PM2.5[a]	HC	CO	SO_2	CO_2
Alaska East (AE)	27,982	2,561	2,356	1,073	2,534	19,084	1,182,047
Alaska West (AW)	89,826	8,118	7,469	3,444	8,112	60,227	3,711,596
East Coast (EC)	391,995	39,003	35,882	16,216	38,382	323,038	18,121,202
Gulf Coast (GC)	232,114	23,403	21,531	9,590	23,628	174,751	10,567,512
Hawaii East (HE)	42,935	4,185	3,850	1,765	4,161	31,075	1,930,172
Hawaii West (HW)	60,409	5,888	5,417	2,483	5,855	43,722	2,715,741
North Pacific (NP)	38,051	3,916	3,603	1,706	3,799	27,807	1,800,743
South Pacific (SP)	208,294	20,148	18,536	8,585	20,686	149,751	9,490,502
Great Lakes (GL)	18,768	1,613	1,484	681.914	1,607	11,993	740,624
Total U.S. Metric Tonnes	**1,110,375**	**108,835**	**100,128**	**45,544**	**108,762**	**841,447**	**50,260,140**

[a] Estimated from PM_{10} using a multiplicative conversion factor of 0.92.

The ECA control case inventories for each of the nine geographic regions and the U.S. domain total are presented in Table 2-44. The regional and total inventories include all emissions within the 200 nm US modeling domain. Controls are applied to all regions included in the proposed ECA.

Table 2-44 Category 3 Vessel Inventories for 2020 Proposed ECA Control Case[a]

U.S. Region	Metric Tonnes per Year						
	NOx	PM10	PM2.5[a]	HC	CO	SO_2	CO_2
Alaska East (AE)	25,978	322	296	1,073	2,534	728	1,124,652
Alaska West (AW)	89,826	8,118	7,469	3,444	8,112	60,227	3,711,596
East Coast (EC)	289,671	5,286	4,863	16,216	38,382	11,514	17,233,800
Gulf Coast (GC)	170,861	3,201	2,945	9,590	23,628	6,255	10,034,946
Hawaii East (HE)	32,952	551	507	1,765	4,161	1,187	1,838,832
Hawaii West (HW)	60,409	5,888	5,417	2,483	5,855	43,722	2,715,741
North Pacific (NP)	29,105	539	496	1,706	3,799	1,076	1,715,210
South Pacific (SP)	150,461	2,753	2,533	8,585	20,686	5,786	9,009,986
Great Lakes (GL)	16,420	207	190	681	1,607	420	704,390
Total U.S. Metric Tonnes	**865,684**	**26,864**	**24,715**	**45,544**	**108,762**	**130,914**	**48,089,152**

[a] This scenario assumes ECA controls apply within 200 nautical miles of all U.S. regions. Alaska West and Hawaii West are not subject to ECA controls.

The fuel consumption by fuel type in the baseline and ECA cases is also presented in Table 2-45.

Table 2-45 Fuel Consumption by Category 3 Vessels in Baseline and ECA Scenarios.

U.S. Region	Baseline Metric Tonnes Fuel			With ECA Metric Tonnes Fuel		
	Distillate	Residual	Total	Distillate	Residual	Total
Alaska East (AE)	3,386	367,977	371,363	353,331	0	353,331
Alaska West (AW)	0	1,166,068	1,166,068	0	1,166,068	1,166,068
East Coast (EC)	202,139	5,490,981	5,693,120	5,414,326	0	5,414,326
Gulf Coast (GC)	96,428	3,223,557	3,319,985	3,152,669	0	3,152,669
Hawaii East (HE)	10,529	595,871	606,400	577,704	0	577,704
Hawaii West (HW)	0	853,202	853,202	0	853,202	853,202
North Pacific (NP)	28,532	537,206	565,738	538,866	0	538,866
South Pacific (SP)	83,576	2,898,045	2,981,622	2,830,658	0	2,830,658
Great Lakes (GL)	1,269	231,412	232,681	221,297	0	221,297
Total U.S. Metric Tonnes	**425,860**	**15,364,319**	**15,790,179**	**13,088,852**	**2,019,270**	**15,108,122**

2.5 Projected Emission Reductions

The projected reduction (tonnes) for the 2020 control case relative to the 2020 baseline is presented in Table 2-46. Reductions by region, for the total U.S., and for the total 48-states, are provided by pollutant in each table.

Table 2-46 Reductions for 2020 Proposed ECA Control Case[a]

U.S. Region	Metric Tonnes per Year						
	NO_X	PM_{10}	$PM_{2.5}$[a]	HC	CO	SO_2	CO_2
Alaska East (AE)	2,004	2,239	2,060	0	0	18,356	57,395
Alaska West (AW)	0	0	0	0	0	0	0
East Coast (EC)	102,324	33,717	31,019	0	0	311,524	887,402
Gulf Coast (GC)	61,253	20,202	18,586	0	0	168,496	532,566
Hawaii East (HE)	9,983	3,634	3,343	0	0	29,888	91,340
Hawaii West (HW)	0	0	0	0	0	0	0
North Pacific (NP)	8,946	3,377	3,107	0	0	26,731	85,533
South Pacific (SP)	57,833	17,395	16,003	0	0	143,965	480,516
Great Lakes (GL)	2,348	1,406	1,294	0	0	11,573	36,234
Total U.S. Metric Tonnes	244,690	81,971	75,413	0	0	710,534	2,170,987

[a] The emission reductions are relative to the 2020 baseline.

2.6 Conclusion

An emission inventory for ships in the U.S. was developed based on the latest state of the art models and inputs, using a “bottom-up” methodology. The inventory includes emissions for 117 ports, as well as emissions for ships while underway in U.S. waters. The analysis clearly demonstrates that emissions from ships in the proposed ECA are contributing to U.S. air pollution. The inventory data were used as an input for the air quality modeling analysis.

Appendices

Appendix 2A: Port Coordinates

Table 2A-1 Port Coordinates

Port Name	US ACE Code	Port Coordinates	
		Longitude	Latitude
Albany, NY	C0505	-73.7482	42.64271
Alpena, MI	L3617	-83.4223	45.0556
Anacortes, WA	C4730	-122.6	48.49617
Anchorage, AK	C4820	-149.895	61.23778
Ashtabula, OH	L3219	-80.7917	41.91873
Baltimore, MD	C0700	-76.5171	39.20899
Barbers Point, Oahu, HI	C4458	-158.109	21.29723
Baton Rouge, LA	C2252	-91.1993	30.42292
Beaumont, TX	C2395	-94.0881	30.08716
Boston, MA	C0149	-71.0523	42.35094
Bridgeport, CT	C0311	-73.1789	41.172
Brownsville, TX	C2420	-97.3981	25.9522
Brunswick, GA	C0780	-81.4999	31.15856
Buffalo, NY	L3230	-78.8953	42.8783
Burns Waterway Harbor, IN	L3739	-87.1552	41.64325
Calcite, MI	L3620	-83.7756	45.39293
Camden-Gloucester, NJ	C0551	-75.1043	39.94305
Carquinez, CA	CCA01	-122.123	38.03556
Catalina, CA	CCA02	-118.496	33.43943
Charleston, SC	C0773	-79.9216	32.78878
Chester, PA	C0297	-75.3222	39.85423
Chicago, IL	L3749	-87.638	41.88662
Cleveland, OH	L3217	-81.6719	41.47852
Conneaut, OH	L3220	-80.5486	41.96671
Coos Bay, OR	C4660	-124.21	43.36351
Corpus Christi, TX	C2423	-97.3979	27.81277
Detroit, MI	L3321	-83.1096	42.26909
Duluth-Superior, MN and WI	L3924	-92.0964	46.77836
El Segundo, CA	CCA03	-118.425	33.91354
Erie, PA	L3221	-80.0679	42.15154
Escanaba, MI	L3795	-87.025	45.73351
Eureka, CA	CCA04	-124.186	40.79528
Everett, WA	C4725	-122.229	47.98476
Fairport Harbor, OH	L3218	-81.2941	41.76666
Fall River, MA	C0189	-71.1588	41.72166
Freeport, TX	C2408	-95.3304	28.9384
Galveston, TX	C2417	-94.8127	29.31049
Gary, IN	L3736	-87.3251	41.61202
Georgetown, SC	C0772	-79.2896	33.36682
Grays Harbor, WA	C4702	-124.122	46.91167
Gulfport, MS	C2083	-89.0853	30.35216
Hilo, HI	C4400	-155.076	19.72861

Port Name	US ACE Code	Port Coordinates	
		Longitude	Latitude
Honolulu, HI	C4420	-157.872	21.31111
Hopewell, VA	C0738	-77.2763	37.32231
Houston, TX	C2012	-95.2677	29.72538
Indiana Harbor, IN	L3738	-87.4455	41.67586
Jacksonville, FL	C2017	-81.6201	30.34804
Kahului, Maui, HI	C4410	-156.473	20.89861
Kalama, WA	C4626	-122.863	46.02048
Lake Charles, LA	C2254	-93.2221	30.22358
Long Beach, CA	C4110	-118.21	33.73957
Longview, WA	C4622	-122.914	46.14222
Lorain, OH	L3216	-82.1951	41.48248
Los Angeles, CA	C4120	-118.241	33.77728
Manistee, MI	L3720	-86.3443	44.25082
Marblehead, OH	L3212	-82.7091	41.52962
Marcus Hook, PA	C5251	-75.4042	39.81544
Matagorda Ship Channel, TX	C2410	-96.5641	28.5954
Miami, FL	C2164	-80.1832	25.78354
Milwaukee, WI	L3756	-87.8997	42.98824
Mobile, AL	C2005	-88.0411	30.72527
Morehead City, NC	C0764	-76.6947	34.71669
Muskegon, MI	L3725	-86.3501	43.19492
Nawiliwili, Kauai, HI	C4430	-159.353	21.96111
New Bedford, MA	C0187	-70.9162	41.63641
New Castle, DE	C0299	-75.5616	39.65668
New Haven, CT	C1507	-72.9047	41.29883
New Orleans, LA	C2251	-90.0853	29.91414
New York, NY and NJ	C0398	-74.0384	40.67395
Newport News, VA	C0736	-76.4582	36.98522
Nikishka, AK	C4831	-151.314	60.74793
Oakland, CA	C4345	-122.308	37.82152
Olympia, WA	C4718	-122.909	47.06827
Other Puget Sound, WA	C4754	-122.72	48.84099
Palm Beach, FL	C2162	-80.0527	26.76904
Panama City, FL	C2016	-84.1993	30.19009
Pascagoula, MS	C2004	-88.5588	30.34802
Paulsboro, NJ	C5252	-75.2266	39.82689
Penn Manor, PA	C0298	-74.7408	40.13598
Pensacola, FL	C2007	-87.2579	30.40785
Philadelphia, PA	C0552	-75.2022	39.91882
Plaquemines, LA, Port of	C2255	-89.6875	29.48
Port Angeles, WA	C4708	-123.453	48.1305
Port Arthur, TX	C2416	-93.9607	29.83142
Port Canaveral, FL	C2160	-80.6082	28.41409
Port Dolomite, MI	L3627	-84.3128	45.99139
Port Everglades, FL	C2163	-80.1178	26.09339
Port Hueneme, CA	C4150	-119.208	34.14824
Port Inland, MI	L3803	-85.8628	45.95508

Port Name	US ACE Code	Port Coordinates	
		Longitude	Latitude
Port Manatee, FL	C2023	-82.5613	27.63376
Portland, ME	C0128	-70.2513	43.64951
Portland, OR	C4644	-122.665	45.47881
Presque Isle, MI	L3845	-87.3852	46.57737
Providence, RI	C0191	-71.3984	41.81178
Redwood City, CA	CCA05	-122.21	37.51306
Richmond, CA	C4350	-122.374	37.92424
Richmond, VA	C0737	-77.4194	37.45701
Sacramento, CA	CCA06	-121.544	38.56167
San Diego, CA	C4100	-117.178	32.70821
San Francisco, CA	C4335	-122.399	37.80667
Sandusky, OH	L3213	-82.7123	41.47022
Savannah, GA	C0776	-81.0954	32.08471
Searsport, ME	C0112	-68.925	44.45285
Seattle, WA	C4722	-122.359	47.58771
South Louisiana, LA, Port of	C2253	-90.6179	30.03345
St. Clair, MI	L3509	-82.4941	42.82663
Stockton, CA	C4270	-121.316	37.9527
Stoneport, MI	L3619	-83.4703	45.28073
Tacoma, WA	C4720	-122.452	47.28966
Tampa, FL	C2021	-82.5224	27.78534
Texas City, TX	C2404	-94.9181	29.36307
Toledo, OH	L3204	-83.5075	41.66294
Two Harbors, MN	L3926	-91.6626	47.00428
Valdez, AK	C4816	-146.346	61.12473
Vancouver, WA	C4636	-122.681	45.62244
Wilmington, DE	C0554	-75.507	39.71589
Wilmington, NC	C0766	-77.954	34.23928

Appendix 2B: Port Methodology and Equations

Near port emissions for each port are calculated for four modes of operation: 1) hotelling, 2) maneuvering, 3) reduced speed zone (RSZ), and 4) cruise. Hotelling, or dwelling, occurs while the vessel is docked or anchored near a dock, and only the auxiliary engine(s) are being used to provide power to meet the ship's energy needs. Maneuvering occurs within a very short distance of the docks. The RSZ varies from port to port, though generally the RSZ would begin and end when the pilots board or disembark, and typically occurs when the near port shipping lanes reach unconstrained ocean shipping lanes. The cruise mode emissions in the near ports analysis extend 25 nautical miles beyond the end of the RSZ lanes for deep water ports and 7 nautical miles for Great Lake ports.

Emissions are calculated separately for propulsion and auxiliary engines. The basic equation used is as follows:

Equation 2B-1

$$Emissions_{mode[eng]} = (calls) \times (P_{[eng]}) \times (hrs/call_{mode}) \times (LF_{mode[eng]}) \times (EF_{[eng]}) \times (Adj) \times (10^{-6}\ tonnes/g)$$

Where:
$Emissions_{mode\ [eng]}$ = Metric tonnes emitted by mode and engine type
Calls = Round-trip visits (i.e., one entrance and one clearance is considered a call)
$P_{[eng]}$ = Total engine power by engine type, in kilowatts
$hrs/call_{mode}$ = Hours per call by mode
$LF_{mode\ [eng]}$ = Load factor by mode and engine type (unitless)
$EF_{[eng]}$ = Emission factor by engine type for the pollutant of interest, in g/kW-hr
(these vary as a function of engine type and fuel used, rather than activity mode)
Adj = Low load adjustment factor, unitless (used when the load factor is below 0.20)
10^{-6} = Conversion factor from grams to metric tonnes

Main engine load factors are calculated directly from the propeller curve based upon the cube of actual speed divided by maximum speed (at 100% maximum continuous rating [MCR]). In addition, cruise mode activity is based on cruise distance and speed inputs. The following sections provide the specific equations used to calculate propulsion and auxiliary emissions for each activity mode.

Cruise

Cruise emissions are calculated for both propulsion (main) and auxiliary engines. The basic equation used to calculate cruise mode emissions for the main engines is:

Equation 2B-2

$$Emissions_{cruise[main]} = (calls) \times (P_{[main]}) \times (hrs/call_{cruise}) \times (LF_{cruise[main]}) \times (EF_{[main]}) \times (10^{-6}\ tonnes/g)$$

Where:
$Emissions_{cruise\ [main]}$ = Metric tonnes emitted from main engines in cruise mode
Calls = Round-trip visits (i.e., one entrance and one clearance is considered a call)

$P_{[main]}$ = Total main engine power, in kilowatts
$hrs/call_{cruise}$ = Hours per call for cruise mode
$LF_{cruise\,[main]}$ = Load factor for main engines in cruise mode (unitless)
$EF_{[main]}$ = Emission factor for main engines for the pollutant of interest, in g/kW-hr (these vary as a function of engine type and fuel used, rather than activity mode)
10^{-6} = Conversion factor from grams to metric tonnes

In addition, the time in cruise is calculated as follows:

Equation 2B-3

$$Hrs / call_{cruise} = Cruise\ Distance\,[nmiles] / Cruise\ Speed\,[knots] \times 2\,trips / call$$

Where:
Cruise distance = one way distance (25 nautical miles for deep sea ports, and 7 nautical miles for Great Lake ports)
Cruise speed = vessel service speed, in knots
2 trips/call = Used to calculate round trip cruise distance

Main engine load factors are calculated directly from the propeller curve based upon the cube of actual speed divided by maximum speed (at 100% maximum continuous rating [MCR]):

Equation 2B-4

$$LoadFactor_{cruise\,[main]} = \left(Cruise\ Speed\,[knots] / Maximum\ Speed\,[knots]\right)^3$$

Since cruise speed is estimated at 94 percent of maximum speed[38], the load factor for main engines at cruise is 0.83.

Substituting Equation 2B-3 for time in cruise into Equation 2B-2, and using the load factor of 0.83, the equation used to calculate cruise mode emissions for the main engines becomes the following:

Equation 2B-5 Cruise Mode Emissions for Main Engines

$$Emissions_{cruise[main]} = (calls) \times (P_{[main]}) \times (CruiseDistance/CruiseSpeed) \times (2\,trips/call) \times 0.83 \times (EF_{[main]}) \times (10^{-6}\ tonne$$

Where:
$Emissions_{cruise\,[main]}$ = Metric tonnes emitted from main engines in cruise mode
calls = Round-trip visits (i.e., one entrance and one clearance is considered a call)
$P_{[main]}$ = Total main engine power, in kilowatts
Cruise distance = one way distance (25 nautical miles for deep sea ports, and 7 nautical miles for Great Lake ports)
Cruise speed = vessel service speed, in knots
2 trips/call = Used to calculate round trip cruise distance
0.83 = Load factor for main engines in cruise mode, unitless
$EF_{[main]}$ = Emission factor for main engines for the pollutant of interest, in g/kW-hr (these vary as a function of engine type and fuel used, rather than activity mode)
10^{-6} = Conversion factor from grams to metric tonnes

The equation used to calculate cruise mode emissions for the auxiliary engines is:

Equation 2B-6 Cruise Mode Emissions for Auxiliary Engines

$$Emissions_{cruise[aux]} = (calls) \times (P_{[aux]}) \times (Cruise\ Distance/Cruise\ Speed) \times (2\, trips/call) \times (LF_{cruise[aux]}) \times (EF_{[aux]}) \times (10^{-6}\ tonnes\ /$$

Where:
$Emissions_{cruise[aux]}$ = Metric tonnes emitted from auxiliary engines in cruise mode
calls = Round-trip visits (i.e., one entrance and one clearance is considered a call)
$P_{[aux]}$ = Total auxiliary engine power, in kilowatts
Cruise distance = one way distance (25 nautical miles for deep sea ports, and 7 nautical miles for Great Lake ports)
Cruise speed = vessel service speed, in knots
2 trips/call = Used to calculate round trip cruise distance
$LF_{cruise\ [aux]}$ = Load factor for auxiliary engines in cruise mode, unitless (these vary by ship type and activity mode)
$EF_{[aux]}$ = Emission factor for auxiliary engines for the pollutant of interest, in g/kW-hr (these vary as a function of engine type and fuel used, rather than activity mode)
10^{-6} = Conversion factor from grams to metric tonnes

The inputs of calls, cruise distance, and vessel speed are the same for main and auxiliary engines. Relative to the main engines, auxiliary engines have separate inputs for engine power, load factor, and emission factors. The activity-related inputs, such as engine power, vessel speed, and calls, can be unique to each ship calling on a port, if ship-specific information is available. For this analysis, these inputs were developed by port for bins that varied by ship type, engine type, and dead weight tonnage (DWT) range.

Reduced Speed Zone

RSZ emissions are calculated for both propulsion (main) and auxiliary engines. The basic equation used to calculate RSZ mode emissions for the main engines is:

Equation 2B-7

$$Emissions_{RSZ[main]} = (calls) \times (P_{[main]}) \times (hrs/call_{RSZ}) \times (LF_{RSZ[main]}) \times (EF_{[main]}) \times (Adj) \times (10^{-6}\ tonnes/g)$$

Where:
$Emissions_{RSZ[main]}$ = Metric tonnes emitted from main engines in RSZ mode
calls = Round-trip visits (i.e., one entrance and one clearance is considered a call)
$P_{[main]}$ = Total main engine power, in kilowatts
$hrs/call_{RSZ}$ = Hours per call for RSZ mode
$LF_{RSZ\ [main]}$ = Load factor for main engines in RSZ mode, unitless
$EF_{[main]}$ = Emission factor for main engines for the pollutant of interest, in g/kW-hr (these vary as a function of engine type and fuel used, rather than activity mode)
Adj = Low load adjustment factor, unitless (used when the load factor is below 0.20)
10^{-6} = Conversion factor from grams to metric tonnes

In addition, the time in RSZ mode is calculated as follows:

Equation 2B-8

$$Hrs\ /\ call_{RSZ} = RSZ\ Distance[nmiles]\ /\ RSZ\ Speed\ [knots] \times 2\ trips\ /\ call$$

Load factor during the RSZ mode is calculated as follows:

Equation 2B-9

$$LoadFactor_{RSZ[main]} = \left(RSZ\ Speed\ /\ Maximum\ Speed\right)^3$$

In addition:

Equation 2B-10

$$Maximum\ Speed = Cruise\ Speed\ /\ 0.94$$

Where:
0.94 = Fraction of cruise speed to maximum speed

Substituting Equation 2B-10 into Equation 2B-9, the equation to calculate load factor becomes:

Equation 2B-11

$$LoadFactor_{RSZ[main]} = \left(RSZ\ Speed \times 0.94\ /\ Cruise\ Speed\right)^3$$

Where:
0.94 = Fraction of cruise speed to maximum speed

Load factors below 2 percent were set to 2 percent as a minimum.

Substituting Equation 2B-8 for time in mode and Equation 2B-11 for load factor into Equation 2B-7 , the expression used to calculate RSZ mode emissions for the main engines becomes:

Equation 2B-12 RSZ Mode Emissions for Main Engines

$$Emissions_{RSZ[aux]} = (calls) \times (P_{[aux]}) \times (RSZ\ Distance/\ RSZ\ Speed) \times (2\ trips/call)$$
$$\times \left(RSZ\ Speed \times 0.94\ /\ Cruise\ Speed\right)^3 \times (EF_{[aux]}) \times \left(Adj\right) \times (10^{-6}\ tonnes\ /\ g)$$

Where:
$Emissions_{RSZ[main]}$ = Metric tonnes emitted from main engines in RSZ mode
calls = Round-trip visits (i.e., one entrance and one clearance is considered a call)
$P_{[main]}$ = Total main engine power, in kilowatts
RSZ distance = one way distance, in nautical miles (specific to each port)
RSZ speed = speed, in knots (specific to each port)
2 trips/call = Used to calculate round trip RSZ distance
Cruise speed = vessel service speed, in knots
$EF_{[main]}$ = Emission factor for main engines for the pollutant of interest, in g/kW-hr (these vary as a function of engine type and fuel used, rather than activity mode)
Adj = Low load adjustment factor, unitless (used when the load factor is below 0.20)

10^{-6} = Conversion factor from grams to tons
0.94 = Fraction of cruise speed to maximum speed

Emission factors are considered to be relatively constant down to about 20 percent load. Below that threshold, emission factors tend to increase significantly as the load decreases. During the RSZ mode, load factors can fall below 20 percent. Low load multiplicative adjustment factors were developed and applied when the load falls below 20 percent (0.20). If the load factor is 0.20 or greater, the low load adjustment factor is set to 1.0.

The equation used to calculate RSZ mode emissions for the auxiliary engines is:

Equation 2B-13 RSZ Mode Emissions for Auxiliary Engines

$$Emissions_{RSZ[aux]} = (calls) \times (P_{[aux]}) \times (RSZ\ Distance / RSZ\ Speed) \times (2\ trips/call) \times (LF_{RSZ[aux]}) \times (EF_{[aux]}) \times (10^{-6}\ tonnes / g)$$

Where:
$Emissions_{RSZ[aux]}$ = Metric tonnes emitted from auxiliary engines in RSZ mode
calls = Round-trip visits (i.e., one entrance and one clearance is considered a call)
$P_{[aux]}$ = Total auxiliary engine power, in kilowatts
RSZ distance = one way distance, in nautical miles (specific to each port)
RSZ speed = speed, in knots (specific to each port)
2 trips/call = Used to calculate round trip cruise distance
$LF_{RSZ\ [aux]}$ = Load factor for auxiliary engines in RSZ mode, unitless (these vary by ship type and activity mode)
$EF_{[aux]}$ = Emission factor for auxiliary engines for the pollutant of interest, in g/kW-hr (these vary as a function of engine type and fuel used, rather than activity mode)
10^{-6} = Conversion factor from grams to metric tonnes

Unlike main engines, there is no need for a low load adjustment factor for auxiliary engines, because of the way they are generally operated. When only low loads are needed, one or more engines are shut off, allowing the remaining engines to maintain operation at a more efficient level.

The inputs of calls, RSZ distance, and RSZ speed are the same for main and auxiliary engines. Relative to the main engines, auxiliary engines have separate inputs for engine power, load factor, and emission factors. The RSZ distances vary by port rather than vessel or engine type. Some RSZ speeds vary by ship type, while others vary by DWT. Mostly, however, RSZ speed is constant for all ships entering the harbor area. All Great Lake ports have reduced speed zone distances of three nautical miles occurring at halfway between cruise speed and maneuvering speed.

Maneuvering

Maneuvering emissions are calculated for both propulsion (main) and auxiliary engines. The basic equation used to calculate maneuvering mode emissions for the main engines is:

Equation 2B-14

$$Emissions_{man[main]} = (calls) \times (P_{[main]}) \times (hrs / call_{man}) \times (LF_{man[main]}) \times (EF_{[main]}) \times (Adj) \times (10^{-6}\ tonnes / g)$$

Where:

$Emissions_{man[main]}$ = Metric tonnes emitted from main engines in maneuvering mode
calls = Round-trip visits (i.e., one entrance and one clearance is considered a call)
$P_{[main]}$ = Total main engine power, in kilowatts
$hrs/call_{man}$ = Hours per call for maneuvering mode
$LF_{man\ [main]}$ = Load factor for main engines in maneuvering mode, unitless
$EF_{[main]}$ = Emission factor for main engines for the pollutant of interest, in g/kW-hr (these vary as a function of engine type and fuel used, rather than activity mode)
Adj = Low load adjustment factor, unitless (used when the load factor is below 0.20)
10^{-6} = Conversion factor from grams to metric tonnes

Maneuvering time-in-mode is estimated based on the distance a ship travels from the breakwater or port entrance to the pier/wharf/dock (PWD). Maneuvering times also include shifts from one PWD to another or from one port within a greater port area to another. Average maneuvering speeds vary from 3 to 8 knots depending on direction and ship type. For consistency, maneuvering speeds were assumed to be the dead slow setting of approximately 5.8 knots.

Load factor during maneuvering is calculated as follows:

Equation 2B-15

$$LoadFactor_{man[main]} = \left(Man\,Speed[knots] / Maximum\,Speed[knots]\right)^3$$

In addition:

Equation 2B-16

$$Maximum\,Speed = Cruise\,Speed[knots] / 0.94$$

Where:
0.94 = Fraction of cruise speed to maximum speed

Also, the maneuvering speed is 5.8 knots. Substituting Equation 2B-16 into Equation 2B-15, and using a maneuvering speed of 5.8 knots, the equation to calculate load factor becomes:

Equation 2B-17

$$LoadFactor_{man[main]} = \left(5.45 / Cruise\,Speed\right)^3$$

Load factors below 2 percent were set to 2 percent as a minimum.

Substituting Equation 2B-17 for load factor into Equation 2B-14, the expression used to calculate maneuvering mode emissions for the main engines becomes:

Equation 2B-18 Maneuvering Mode Emissions for Main Engines

$$Emissions_{man[main]} = (calls) \times (P_{[main]}) \times (hrs / call_{man}) \times (5.45 / Cruise\ Speed)^3 \times (EF_{[main]}) \times (Adj) \times (10^{-6}\ tonnes\ /\ g)$$

Where:
$Emissions_{man[main]}$ = Metric tonnes emitted from main engines in maneuvering mode
calls = Round-trip visits (i.e., one entrance and one clearance is considered a call)
$P_{[main]}$ = Total main engine power, in kilowatts

$hrs/call_{man}$ = Hours per call for maneuvering mode
Cruise speed = Vessel service speed, in knots
$EF_{[main]}$ = Emission factor for main engines for the pollutant of interest, in g/kW-hr (these vary as a function of engine type and fuel used, rather than activity mode)
Adj = Low load adjustment factor, unitless (used when the load factor is below 0.20)
10^{-6} = Conversion factor from grams to metric tonnes

Since the load factor during maneuvering usually falls below 20 percent, low load adjustment factors are also applied accordingly. Maneuvering times are not readily available for all 117 ports. For this analysis, maneuvering times and load factors available for a subset of the ports were used to calculate maneuvering emissions for the remaining ports. This is discussed in more detail in section 2.3.2.3.8.

The equation used to calculate maneuvering mode emissions for the auxiliary engines is:

Equation 2B-19 Maneuvering Mode Emissions for Auxiliary Engines

$$Emissions_{man[aux]} = (calls) \times (P_{[aux]}) \times (hrs / call_{man}) \times (LF_{man[aux]}) \times (EF_{[aux]}) \times (10^{-6}\ tonnes / g)$$

Where:
$Emissions_{man[aux]}$ = Metric tonnes emitted from auxiliary engines in maneuvering mode
calls = Round-trip visits (i.e., one entrance and one clearance is considered a call)
$P_{[aux]}$ = Total auxiliary engine power, in kilowatts
$hrs/call_{man}$ = Hours per call for maneuvering mode
$LF_{man\ [aux]}$ = Load factor for auxiliary engines in maneuvering mode, unitless (these vary by ship type and activity mode)
$EF_{[aux]}$ = Emission factor for auxiliary engines for the pollutant of interest, in g/kW-hr (these vary as a function of engine type and fuel used, rather than activity mode)
10^{-6} = Conversion factor from grams to metric tonnes

Low load adjustment factors are not applied for auxiliary engines.

Hotelling

Hotelling emissions are calculated for auxiliary engines only, as main engines are not operational during this mode. The equation used to calculate hotelling mode emissions for the auxiliary engines is:

Equation 2B-20 Hotelling Mode Emissions for Auxiliary Engines

$$Emissions_{hotel[aux]} = (calls) \times (P_{[aux]}) \times (hrs / call_{hotel}) \times (LF_{hotel[aux]}) \times (EF_{[aux]}) \times (10^{-6}\ tonnes / g)$$

Where:
$Emissions_{hotel[aux]}$ = Metric tonnes emitted from auxiliary engines in hotelling mode
calls = Round-trip visits (i.e., one entrance and one clearance is considered a call)
$P_{[aux]}$ = Total auxiliary engine power, in kilowatts
$hrs/call_{hotel}$ = Hours per call for hotelling mode
$LF_{hotel\ [aux]}$ = Load factor for auxiliary engines in hotelling mode, unitless (these vary by ship type and activity mode)

$EF_{[aux]}$ = Emission factor for auxiliary engines for the pollutant of interest, in g/kW-hr (these vary as a function of engine type and fuel used, rather than activity mode)
10^{-6} = Conversion factor from grams to metric tonnes

Hotelling times are not readily available for all 117 ports. For this analysis, hotelling times available for a subset of the ports were used to calculate hotelling emissions for the remaining ports.

Appendix 2C: Port Reduced Speed Zone (RSZ) Information

Table 2C-1 Port RSZ Information

Port Name	RSZ Speed (knts)	RSZ distance (naut mi)	Final RSZ End Point(s)	
			Longitude	Latitude
Albany, NY	c	142.5	-73.8929	40.47993
Alpena, MI	e	3	-83.2037	44.99298
Anacortes, WA	a	108.3	-124.771	48.49074
Anchorage, AK	14.5	143.6	-152.309	59.5608
Ashtabula, OH	e	3	-80.8097	42.08549
Baltimore, MD	c	157.1	-75.8067	36.8468
Barbers Point, Oahu, HI	10	5.1	-158.132	21.21756
			-89.4248	28.91161
Baton Rouge, LA	10	219.8	-89.137	28.98883
Beaumont, TX	7	53.5	-93.7552	29.55417
Boston, MA	10	14.3	-70.7832	42.37881
Bridgeport, CT	10	2	-73.1863	41.13906
Brownsville, TX	8.8	18.7	-97.0921	26.06129
			-80.9345	31.29955
Brunswick, GA	13	38.8	-81.1357	30.68935
Buffalo, NY	e	3	-79.0996	42.81683
Burns Waterway Harbor, IN	e	3	-87.1032	41.80625
Calcite, MI	e	3	-83.5383	45.39496
Camden-Gloucester, NJ	c	94	-75.0095	38.79004
Carquinez, CA	12	39	-122.632	37.76094
Catalina, CA	12	11.9	-118.465	33.63641
Charleston, SC	12	17.3	-79.6452	32.62557
Chester, PA	c	78.2	-75.0095	38.79004
Chicago, IL	e	3	-87.4141	41.86971
Cleveland, OH	e	3	-81.765	41.63079
Conneaut, OH	e	3	-80.5639	42.13361
Coos Bay, OR	6.5	13	-124.359	43.35977
Corpus Christi, TX	d	30.1	-96.8753	27.74433
Detroit, MI	e	3	-83.1384	42.10308
Duluth-Superior, MN and WI	e	3	-91.8536	46.78916
			-118.926	33.91252
El Segundo, CA	12	23.3	-118.465	33.63641
Erie, PA	e	3	-80.115	42.3151
Escanaba, MI	e	3	-86.9224	45.58297
Eureka, CA	12	9	-124.347	40.75925
Everett, WA	a	123.3	-124.771	48.49074
Fairport Harbor, OH	e	3	-81.3917	41.91401
Fall River, MA	9	22.7	-71.3334	41.41708
Freeport, TX	c	2.6	-95.2949	28.93323
Galveston, TX	c	9.3	-94.6611	29.3247
Gary, IN	e	3	-87.2824	41.77658
Georgetown, SC	12	17.6	-79.0779	33.1924

Port Name	RSZ Speed (knts)	RSZ distance (naut mi)	Final RSZ End Point(s)	
			Longitude	Latitude
Grays Harbor, WA	a	4.9	-124.24	46.89509
Gulfport, MS	10	17.4	-88.9263	30.11401
Hilo, HI	10	7.1	-154.985	19.76978
			-157.956	21.17658
Honolulu, HI	10	10	-157.785	21.23827
Hopewell, VA	10	91.8	-75.8067	36.8468
Houston, TX	c	49.6	-94.6611	29.3247
Indiana Harbor, IN	e	3	-87.4007	41.8401
Jacksonville, FL	10	18.6	-81.3649	30.39769
Kahului, Maui, HI	10	7.5	-156.44	21.01066
Kalama, WA	b	68.2	-124.137	46.22011
Lake Charles, LA	6	38	-93.3389	29.73094
			-118.465	33.63641
Long Beach, CA	12	18.1	-118.13	33.45211
Longview, WA	b	67.3	-124.137	46.22011
Lorain, OH	e	3	-82.2701	41.64023
			-118.465	33.63641
Los Angeles, CA	12	20.6	-118.13	33.45211
Manistee, MI	e	3	-86.3819	44.41573
Marblehead, OH	e	3	-82.7293	41.69638
Marcus Hook, PA	c	94.7	-75.0095	38.79004
Matagorda Ship Channel, TX	7.3	24	-96.2287	28.33472
Miami, FL	12	3.8	-80.1201	25.75787
Milwaukee, WI	e	3	-87.6718	42.97343
Mobile, AL	11	36.1	-88.0644	30.1457
Morehead City, NC	10	2.2	-76.6679	34.68999
Muskegon, MI	e	3	-86.5377	43.29151
Nawiliwili, Kauai, HI	10	7.3	-159.266	21.87705
New Bedford, MA	9	22.4	-71.1013	41.38499
New Castle, DE	c	60.5	-75.0095	38.79004
New Haven, CT	10	2.1	-72.9121	41.26588
			-89.4248	28.91161
New Orleans, LA	10	104.2	-89.137	28.98883
New York, NY and NJ	c	15.7	-73.8929	40.47993
Newport News, VA	14	24.3	-75.8067	36.8468
Nikishka, AK	14.5	90.7	-152.309	59.5608
Oakland, CA	12	18.4	-122.632	37.76094
Olympia, WA	a	185.9	-124.771	48.49074
Other Puget Sound, WA	a	106	-124.771	48.49074
Palm Beach, FL	3	3.1	-79.9973	26.77129
Panama City, FL	10	10	-84.1797	30.0818
Pascagoula, MS	10	17.5	-88.4804	30.09597
Paulsboro, NJ	c	83.5	-75.0095	38.79004
Penn Manor, PA	c	114.5	-75.0095	38.79004
Pensacola, FL	12	12.7	-87.298	30.27777
Philadelphia, PA	c	88.1	-75.0095	38.79004

Port Name	RSZ Speed (knts)	RSZ distance (naut mi)	Final RSZ End Point(s)	
			Longitude	Latitude
Plaquemines, LA, Port of	10	52.4	-89.4248	28.91161
			-89.137	28.98883
Port Angeles, WA	a	65	-124.771	48.49074
Port Arthur, TX	7	21	-93.7552	29.55417
Port Canaveral, FL	10	4.4	-80.5328	28.41439
Port Dolomite, MI	e	3	-84.2445	45.83181
Port Everglades, FL	7.5	2.1	-80.082	26.08627
Port Hueneme, CA	12	2.8	-119.238	34.10859
Port Inland, MI	e	3	-85.6524	45.87553
Port Manatee, FL	9	27.4	-83.0364	27.59078
Portland, ME	10	11.4	-70.1077	43.54224
Portland, OR	b	105.1	-124.137	46.22011
Presque Isle, MI	e	3	-87.082	46.5804
Providence, RI	9	24.9	-71.3334	41.41708
Redwood City, CA	12	36	-122.632	37.76094
Richmond, CA	12	22.6	-122.632	37.76094
Richmond, VA	10	106.4	-75.8067	36.8468
Sacramento, CA	12	90.5	-122.632	37.76094
San Diego, CA	12	11.7	-117.315	32.62184
San Francisco, CA	12	14.4	-122.632	37.76094
Sandusky, OH	e	3	-82.5251	41.56193
Savannah, GA	13	45.5	-78.0498	33.83598
Searsport, ME	9	22.2	-68.7645	44.1179
Seattle, WA	a	133.3	-124.771	48.49074
South Louisiana, LA, Port of	10	142.8	-89.4248	28.91161
			-89.137	28.98883
St. Clair, MI	e	3	-82.5838	42.55923
Stockton, CA	12	86.9	-122.632	37.76094
Stoneport, MI	e	3	-83.2355	45.25919
Tacoma, WA	a	150.5	-124.771	48.49074
Tampa, FL	9	30	-83.0364	27.59078
Texas City, TX	c	15.1	-94.6611	29.3247
Toledo, OH	e	3	-83.3034	41.7323
Two Harbors, MN	e	3	-91.4414	46.93391
Valdez, AK	10	27.2	-146.881	60.86513
Vancouver, WA	b	95.7	-124.137	46.22011
Wilmington, DE	c	65.3	-75.0095	38.79004
Wilmington, NC	10	27.6	-80.325	31.84669

[a] Cruise speed through Strait of Juan de Fuca, then varies by ship type for remaining journey
[b] Inbound on Columbia River at 6.5 knots, outbound at 12 knots
[c] Speed varies by ship type similar to typical like port
[d] Speed varies by ship DWTs
[e] All Great Lake ports have reduced speed zone distances of 3 nautical miles with speeds halfway between service speed and maneuvering speed.

Appendix 2D: Use of Detailed Typical Port Data for Other Inputs

There is currently not enough information to readily calculate time-in-mode (hours/call) for all 117 ports during the maneuvering and hotelling modes of operation. As a result, it was necessary to review and select available detailed emission inventories that have been estimated for selected ports to date. These ports are referred to as typical ports. The typical port information for maneuvering and hotelling time-in-mode (as well as maneuvering load factors for the propulsion engines) was then used for the typical ports and also assigned to the other modeled ports. A modeled port is the port in which emissions are to be estimated. The methodology that was used to select the typical ports and match these ports to the other modeled ports is briefly described in this appendix, and more fully described in the ICF documentation.[39]

2.6.1 Selection of Typical Ports

In 1999, the U.S. Government published two guidance documents[40,41] to calculate marine vessel activity at ports. These documents contained detailed port inventories of eight deep sea ports, two Great Lake ports and two inland river ports. The detailed inventories were developed by obtaining ship call data from Marine Exchanges/Port Authorities (MEPA) at the various ports for 1996 and matching the various ship calls to data from Lloyds Maritime Information Services to provide ship characteristics. The ports for which detailed inventories were developed are shown in Table 2D-1 for deep sea ports and Table 2D-2 for Great Lake ports along with the level of detail of shifts for each port. Most ports provided the ship name, Lloyd's number, the vessel type, the date and time the vessel entered and left the port, and the vessel flag. Inland river ports were developed from US Army Corps of Engineers (USACE) Waterborne Commerce Statistics Center data.

Table 2D-1 Deep Sea MEPA Vessel Movement and Shifting Details

MEPA Area and Ports	MEPA Data Includes
Lower Mississippi River including the ports of New Orleans, South Louisiana, Plaquemines, and Baton Rouge	Information on the first and last pier/wharf/dock (PWD) for the vessel (gives information for at most one shift per vessel). No information on intermediate PWDs, the time of arrival at the first destination PWD, or the time of departure from the River.
Consolidated Port of New York and New Jersey and other ports on the Hudson and Elizabeth Rivers	All PWDs or anchorages for shifting are named. Shifting arrival and departure times are not given. Hotelling time is based upon the entrance and clearance times and dates, subtracting out maneuvering times. Maneuvering times were calculated based upon the distance the ship traveled at a given maneuvering speed.
Delaware River Ports including the ports of Philadelphia, Camden, Wilmington and others	All PWDs or anchorages for shifting are named. Shifting arrival and departure times are not given. Hotelling time is based upon the entrance and clearance times and dates, subtracting out maneuvering times. Maneuvering times were calculated based upon the distance the ship traveled at a given maneuvering speed.
Puget Sound Area Ports including the ports of Seattle, Tacoma, Olympia, Bellingham, Anacortes, and Grays Harbor	All PWDs or anchorages for shifting are named. Arrival and departure dates and times are noted for all movements, allowing calculation of maneuvering and hotelling both for individual shifts and the overall call on port.
The Port of Corpus Christi, TX	Only has information on destination PWD and date and time in and out of the port area. No shifting details.

MEPA Area and Ports	MEPA Data Includes
The Port of Coos Bay, OR	Only has information on destination PWD and date and time in and out of the port area. No shifting details.
Patapsco River Ports including the port of Baltimore Harbor, MD	All PWDs or anchorages for shifting are named. Shifting arrival and departure times are not given. Hotelling time is based upon the entrance and clearance times and dates, subtracting out maneuvering times. Maneuvering times were calculated based upon the distance the ship traveled at a given maneuvering speed.
The Port of Tampa, FL	All PWDs or anchorages for shifting are named. Arrival and departure dates and times are noted for all movements, allowing calculation of maneuvering and hotelling both for individual shifts and the overall call.

Table 2D-2 Great Lake MEPA movements and shifts

MEPA Area and Ports	MEPA Data Includes
Port of Cleveland, OH	Information on the first and last PWD for the vessel (gives information for at most one shift per vessel). No information on intermediate PWDs..
Port of Burns Harbor, IN	No shifting details, No PWDs listed..

Since 1999, several new detailed emissions inventories have been developed and were reviewed for use as additional or replacement typical ports: These included:

- Port of Los Angeles[38,42]
- Puget Sound Ports[43]
- Port of New York/New Jersey[44]
- Port of Houston/Galveston[45]
- Port of Beaumont/Port Arthur[46]
- Port of Corpus Christi[47]
- Port of Portland[48]
- Ports of Cleveland, OH and Duluth-Superior, MN&WI[49]

Based on the review of these newer studies, some of the previous typical ports were replaced with newer data and an additional typical port was added. Data developed for Cleveland and Duluth-Superior for LADCO was used in lieu of the previous typical port data for Cleveland and Burns Harbor because it provided more detailed information and better engine category definitions. The Port of Houston/Galveston inventory provided enough data to add an additional typical port. All three port inventories were adjusted to reflect the current methodology used in this study.

The information provided in the current inventory for Puget Sound Ports[43] was used to calculate RSZ speeds, load factors, and times for all Puget Sound ports. As described in Section

2.6.3.2, an additional modeled port was also added to account for the considerable amount of Jones Act tanker ship activity in the Puget Sound area that is not contained in the original inventory.

The newer Port of New York/New Jersey inventory provided a check against estimates made using the 1996 data. All other new inventory information was found to lack sufficient detail to prepare the detailed typical port inventories needed for this project.

The final list of nine deep sea and two Great Lake typical ports used in this analysis and their data year is as follows:

- Lower Mississippi River Ports [1996]
- Consolidated Ports of New York and New Jersey and Hudson River [1996]
- Delaware River Ports [1996]
- Puget Sound Area Ports [1996]
- Corpus Christi, TX [1996]
- Houston/Galveston Area Ports [1997]
- Ports on the Patapsco River [1996]
- Port of Coos Bay, OR [1996]
- Port of Tampa, FL [1996]
- Port of Cleveland, OH on Lake Erie [2005]
- Duluth-Superior, MN & WI on Lake Michigan [2005]

The maneuvering and hotelling time-in-modes, as well as the maneuvering load factors for these typical ports, were binned by ship type, engine type, and DWT type, using the same bins described in the section entitled “Bins by Ship Type, Engine Type, and DWT Range.”

2.6.2 Matching Typical Ports to Modeled Ports

The next step in the process was to match the ports to be modeled with the typical port which was most like it. Three criteria were used for matching a given port to a typical port: regional differences[D], maximum vessel draft, and the ship types that call on a specific port. One container port, for instance, may have much smaller bulk cargo and reefer ships number of calls on that port than another. Using these three criteria and the eleven typical ports that are suitable for port matching, the 89 deep sea ports and 28 Great Lake ports were matched to the typical ports. For a typical port, the modeled and typical port is the same (i.e., the port simply represents itself). For California ports, we used data provided by ARB as discussed in Section 2.6.3. The matched ports for the deep sea ports are provided in Table 2D-3.

[D] The region in which a port was located was used to group top ports as it was considered a primary influence on the characteristics (size and installed power) of the vessels calling at those ports.

Table 2D-3 Matched Ports for the Deep Sea Ports

Modeled Port Name	Typical Like Port
Anacortes, WA	Puget Sound
Barbers Point, HI	Puget Sound
Everett, WA	Puget Sound
Grays Harbor, WA	Puget Sound
Honolulu, HI	Puget Sound
Kalama, WA	Puget Sound
Longview, WA	Puget Sound
Olympia, WA	Puget Sound
Port Angeles, WA	Puget Sound
Portland, OR	Puget Sound
Seattle, WA	Puget Sound
Tacoma, WA	Puget Sound
Vancouver, WA	Puget Sound
Valdez, AK	Puget Sound
Other Puget Sound	Puget Sound
Anchorage, AK	Coos Bay
Coos Bay, OR	Coos Bay
Hilo, HI	Coos Bay
Kahului, HI	Coos Bay
Nawiliwili, HI	Coos Bay
Nikishka, AK	Coos Bay
Beaumont, TX	Houston
Freeport, TX	Houston
Galveston, TX	Houston
Houston, TX	Houston
Port Arthur, TX	Houston
Texas City, TX	Houston
Corpus Christi, TX	Corpus Christi
Lake Charles, LA	Corpus Christi
Mobile, AL	Corpus Christi
Brownsville, TX	Tampa
Gulfport, MS	Tampa
Manatee, FL	Tampa
Matagorda Ship	Tampa
Panama City, FL	Tampa
Pascagoula, MS	Tampa
Pensacola, FL	Tampa
Tampa, FL	Tampa
Everglades, FL	Tampa
New Orleans, LA	Lower Mississippi
Baton Rouge, LA	Lower Mississippi

Modeled Port Name	Typical Like Port
South Louisiana, LA	Lower Mississippi
Plaquemines, LA	Lower Mississippi
Albany, NY	New York/New Jersey
New York/New Jersey	New York/New Jersey
Portland, ME	New York/New Jersey
Georgetown, SC	Delaware River
Hopewell, VA	Delaware River
Marcus Hook, PA	Delaware River
Morehead City, NC	Delaware River
Paulsboro, NJ	Delaware River
Chester, PA	Delaware River
Fall River, MA	Delaware River
New Castle, DE	Delaware River
Penn Manor, PA	Delaware River
Providence, RI	Delaware River
Brunswick, GA	Delaware River
Canaveral, FL	Delaware River
Charleston, SC	Delaware River
New Haven, CT	Delaware River
Palm Beach, FL	Delaware River
Bridgeport, CT	Delaware River
Camden, NJ	Delaware River
Philadelphia, PA	Delaware River
Wilmington, DE	Delaware River
Wilmington, NC	Delaware River
Richmond, VA	Delaware River
Jacksonville, FL	Delaware River
Miami, FL	Delaware River
Searsport, ME	Delaware River
Boston, MA	Delaware River
New Bedford/Fairhaven, MA	Delaware River
Baltimore, MD	Patapsco River
Newport News, VA	Patapsco River
Savannah, GA	Patapsco River
Catalina, CA	ARB Supplied
Carquinez, CA	ARB Supplied
El Segundo, CA	ARB Supplied
Eureka, CA	ARB Supplied
Hueneme, CA	ARB Supplied
Long Beach, CA	ARB Supplied
Los Angeles, CA	ARB Supplied
Oakland, CA	ARB Supplied

Modeled Port Name	Typical Like Port
Redwood City, CA	ARB Supplied
Richmond, CA	ARB Supplied
Sacramento, CA	ARB Supplied
San Diego, CA	ARB Supplied
San Francisco, CA	ARB Supplied
Stockton, CA	ARB Supplied

Great Lake ports were matched to either Cleveland or Duluth as shown in Table 2D-4.

Table 2D-4 Great Lake Match Ports

Port Name	Typical Like Port
Alpena, MI	Cleveland
Buffalo, NY	Cleveland
Burns Waterway, IN	Cleveland
Calcite, MI	Cleveland
Cleveland, OH	Cleveland
Dolomite, MI	Cleveland
Erie, PA	Cleveland
Escanaba, MI	Cleveland
Fairport, OH	Cleveland
Gary, IN	Cleveland
Lorain, OH	Cleveland
Marblehead, OH	Cleveland
Milwaukee, WI	Cleveland
Muskegon, MI	Cleveland
Presque Isle, MI	Cleveland
St Clair, MI	Cleveland
Stoneport, MI	Cleveland
Two Harbors, MN	Cleveland
Ashtabula, OH	Duluth-Superior
Chicago, IL	Duluth-Superior
Conneaut, OH	Duluth-Superior
Detroit, MI	Duluth-Superior
Duluth-Superior, MN&WI	Duluth-Superior
Indiana, IN	Duluth-Superior
Inland Harbor, MI	Duluth-Superior
Manistee, MI	Duluth-Superior
Sandusky, OH	Duluth-Superior
Toledo, OH	Duluth-Superior

Once a modeled port was matched to a typical port, the maneuvering and hotelling time-in-mode values, as well as the maneuvering load factors by bin for the typical ports, were used directly for the modeled ports, with no adjustments.

2.6.2.1 Bin Mismatches

In some cases, the specific DWT range bin at the modeled port was not in the typical like port data. In those cases, the next nearest DWT range bin was used for the calculations. In a few cases, the engine type for a given ship type might not be in the typical like port data. In these cases, the closest engine type at the typical like port was used. Also in a few cases, a specific ship type in the modeled port data was not in the typical like port data. In this case, the nearest like ship type at the typical port was chosen to calculate emissions at the modeled port.

2.6.3 Stand Alone Ports

In a few cases, the USACE entrances and clearances data was not used to calculate emissions at the modeled port. These include the California ports for which we received data from ARB, the Port of Valdez, Alaska, and a conglomerate port within the Puget Sound area, as described below.

2.6.3.1 California Ports

The California Air Resources Board (ARB) supplied inventories for 14 California ports for 2002. The data received from ARB for the California ports were modified to provide consistent PM and SO_2 emissions to those calculated in this report. In addition, cruise and RSZ emissions were calculated directly based upon average ship power provided in the ARB methodology document[50] and number of calls, because ARB did not calculate cruise emissions, and transit (RSZ) emissions were allocated to counties instead of ports. ARB provided transit distances for each port to calculate the RSZ emissions. Ship propulsion and auxiliary engine power were calculated based upon the methodology previously described for use in computing cruise and RSZ emissions. For maneuvering and hotelling emissions, the ARB values were used and adjusted as discussed below. The data supplied by ARB included domestic traffic as well as foreign cargo traffic.

For PM emission calculations, ARB used an emission factor of 1.5 g/kWh to calculate total PM emissions and factors of 0.96 and 0.937 to convert total PM to PM_{10} and $PM_{2.5}$ respectively. Since an emission factor of 1.4 g/kWh was used in our calculations for PM_{10} and an emission factor of 1.3 g/kWh for $PM_{2.5}$, ARB PM_{10} and $PM_{2.5}$ emissions were multiplied by factors of 0.972 and 0.925, respectively to get consistent PM_{10} and $PM_{2.5}$ emissions for propulsion engines.

For auxiliary engines, ARB used the same emission factors as above, while we used PM_{10} and $PM_{2.5}$ emission factors of 1.3 and 1.2 g/kWh, respectively for passenger ships and 1.1 and 1.0 g/kWh, respectively for all other ships. In the ARB inventory, all passenger ships are treated as electric drive and all emissions are allocated to auxiliary engines. ARB auxiliary engine emissions were thus multiplied by factors of 0.903 and 0.854 respectively for passenger ships and 0.764 and 0.711 respectively for other ships to provide consistent PM emission calculations.

SO_2 emissions were also different between the ARB and these analyses. ARB used a composite[E] propulsion engine SO_2 emission factor of 10.55 g/kWh while we used a composite SO_2 emission factor of 9.57 g/kWh. Thus, ARB SO_2 propulsion emissions were multiplied by a factor of 0.907 to be consistent with our emission calculations. For auxiliary engines, ARB used SO_2 emission factors of 11.48 and 9.34 g/kWh, respectively for passenger and other ships, while we use emission factors of 9.93 and 9.07 g/kWh, respectively. Thus, ARB auxiliary SO_2 emissions were multiplied by factors of 0.865 and 0.971, respectively for passenger and other ships to provide consistent SO_2 emissions.

2.6.3.2 Port in Puget Sound

In the newest Puget Sound inventory[43], it was found that a considerable amount of tanker ships stop at Cherry Point, Ferndale, March Point and other areas which are not within the top 89 U.S. deep sea ports analyzed in this analysis. In addition, since they are ships carrying U.S. cargo (oil from Alaska) from one U.S. port to another, they are not documented in the USACE entrances and clearances data. To compensate for this anomaly, an additional port was added which encompassed these tanker ships stopping within the Puget Sound area but not at one of the Puget Sound ports analyzed in this analysis. Ship calls in the 1996 typical port data to ports other than those in the top 89 U.S. deep sea ports were analyzed separately. There were 363 ship calls by tankers to those areas in 1996. In the inventory report for 2005, there were 468 calls. For 2002, it was estimated there were 432 calls. The same ship types and ship characteristics were used as in the 1996 data, but the number of calls was proportionally increased to 432 calls to represent these ships. The location of the "Other Puget Sound" port was approximately at Cherry Point near Aberdeen.

2.6.3.3 Port of Valdez

In a recent Alaska port inventory,[51] it was found that significant Category 3 domestic tanker traffic enters and leaves the Port of Valdez on destination to West Coast ports. Since the USACE entrances and clearances data did not contain any tanker calls at Valdez in 2002, the recent Alaska inventory data was used to calculate emissions at that port. In this case, the number of calls and ship characteristics for 2002 were taken directly from the Alaska inventory and used in determining emissions for the modeled port with the Puget Sound area typical port being used as the like port.

[E] Based upon ARB assuming 95 percent of the engines were SSD and 5 percent were MSD. The composite SO_2 EF of 9.57 g/kW-hr was calculated using this weighting, along with the SSD and MSD SO_2 EFs for the West Coast ports reported in Table 2-4.

Appendix 2E: Emission Inputs to STEEM

The STEEM waterway network model relies on a number of inputs to identify the movements for each vessel, individual ship attributes, and related emission factor information. Each of these databases is described separately below.

2.6.4 Shipping Movements

The shipping activity and routes database provides information on vessel movements or trips. It is developed using port entrances and clearances information from the USACE report for the U.S. and the Lloyd's Maritime Intelligence Unit (LMIU) for Canada and Mexico.[52] These sources contain information for each vessel carrying foreign cargo at each major port or waterway that, most importantly for this analysis, includes:

- Vessel name
- Last port of call (entrance record) or next port of call (clearance record)

The database then establishes unique identification numbers for each ship, each port pair, and each resulting trip.

2.6.5 Ship Attributes

The ship attributes data set contains the important characteristics of each ship that are necessary for the STEEM interport model to calculate the emissions associated with each trip. The information in this data set is matched to each previously assigned ship identification number. The following information comes from the USACE entrances and clearances report for each ship identification number:

- Ship type
- Gross registered tonnage (GRT)
- Net registered tonnage (NRT)

The ship attributes data set contains the following information from Lloyd's Register-Fairplay for each ship identification number.

- Main propulsion engine installed power (horsepower)
- Service speed (cruise speed)
- Ship size (length, wide, and draft)

Sometimes data was lacking from the above references for ship speed. In these instances, the missing information was developed for each of nine vessel types and the appropriate value was applied to each individual ship of that type. Specifically, the missing ship speeds for each ship category were obtained from the average speeds used in a Lloyd's Register study of the Baltic Sea and from an Entec UK Limited study for the European Commission.[53,54] The resulting vessel cruise speeds for ships with missing data are shown in Table 2E-1.

Table 2E-1 Average Vessel Cruise Speed by Ship Type [a]

Ship Type	Average Cruise Speed (knots)
Bulk Carrier	14.1
Container Ship	19.9
General Cargo	12.3
Passenger Ship	22.4
Refrigerated Cargo	16.4
Roll On-Roll Off	16.9
Tanker	13.2
Fishing	11.7
Miscellaneous	12.7

[a] Used only when ship specific data were missing from the commercial database references.

The average speed during maneuvering is approximately 60 percent of a ship's cruise speed based on using the propeller law described earlier and the engine load factor for maneuvering that is presented later in this section.

As with vessel cruise speed, main engine installed power was sometimes lacking in the Lloyd's Register-Fairplay data set. Here again, the missing information was developed for nine different vessel types and the appropriate value was applied to each individual ship of that type when the data were lacking. In this case, the missing main engine horsepower was estimated by regressing the relationships between GRT and NRT, and between installed power and GRT for each category. This operation is performed internally in the model and the result applied to each individual ship, as appropriate.

The ship attributes database also contains information on the installed power of engines used for auxiliary purposes. However, this information is usually lacking in the Lloyds data set, so an alternative technique was employed to estimate the required values. In short, the STEEM model uses a ratio of main engine horsepower to auxiliary engine horsepower that was determined for eight different vessel types using information primarily from ICF International.[55] (The ICF report attributed these power values to a study for the Port of Los Angeles by Starcrest Consulting.[38]) The auxiliary engine power for each individual vessel of a given ship type is then estimated by multiplying the appropriate main power to auxiliary power ratio and the main engine horsepower rating for that individual ship. The main and auxiliary power values and the resulting auxiliary engine to main engine ratios are shown in Table 2E-2.

Table 2E-2 Auxiliary Engine Power Ratios

Vessel Type	Average Main Engine Power (kW)	Average Auxiliary Engine Power (kW)	*Auxiliary to Main Engine Power Ratio*
Bulk Carrier	7,954	1,169	*0.147*
Container Ship	30,885	5,746	*0.186*
General Cargo	9,331	1,777	*0.190*
Passenger Ship	39,563	39,563 [a]	*1.000*
Refrigerated Cargo	9,567	3,900 [b]	*0.136*
Roll On-Roll Off	10,696 [c]	2,156 [c]	*0.202*
Tanker	9,409	1,985	*0.211*
Miscellaneous	*6,252*	*1,680*	*0.269*

[a] The ICF reference reported a value of 11,000 for auxiliary engines used on passenger vessels.[55]

[b] The STEEM used auxiliary engine power as reported in the ARB methodology document.[50]

[c] The STEEM purportedly used values for Roll On-Roll Off main and auxiliary engines that represent a trip weighted average of the Auto Carrier and Cruise Ship power values from the ICF reference.

Finally, the ship attributes database provides information on the load factors for main engines during cruise and maneuvering operation, in addition to load factors for auxiliary marine engines. Main engine load factors for cruise operation were taken from a study of international shipping for all ship types, except passenger vessels.[56] For this analysis, the STEEM model used a propulsion engine load factor for passenger ship engines at cruise speed of 55 percent of the total installed power. This is based on engine manufacturer data contained in two global shipping studies.[56,57] During maneuvering, it was assumed that all main engines, including those for passenger ships, operate at 20 percent of the installed power. This is consistent with a study done by Entec UK for the European Commission. The main engine load factors at cruise speed by ship type are shown in Table 2E-3.

Auxiliary engine load factors, except for passenger ships, were obtained from the ICF International study referenced above. These values are also shown in Table 2E-. For cruise mode, neither port nor interport portions of the inventory were adjusted for low load operation, as the low load adjustments are only applied to propulsion engines with load factors below 20%.

Table 2E-3 Main and Auxiliary Engine Load Factors at Cruise Speed by Ship Type

Ship Type	Average Main Engine Load Factor (%)	Average Auxiliary Engine Load Factor (%)
Bulk Carrier	75	17
Container Ship	80	13
General Cargo	80	17
Passenger Ship	55	25
Refrigerated Cargo	80	20
Roll On-Roll Off	80	15
Tanker	75	13
Miscellaneous	70	17

2.6.6 Emission Factor Information

The emission factor data set contains emission rates for the various pollutants in terms of grams of pollutant per kilowatt-hour (g/kW-hr). The main engine emission factors are shown in Table 2E-4. The speed specific factors for NO_X, HC, and SO_2 were taken from several recent analyses of ship emissions in the U.S., Canada, and Europe.50'55'56' [58] The PM factor was based on discussions with the California Air Resources Board (ARB) staff. The fuel specific CO emission factor was taken from a report by ENVIRON International.[59] The STEEM study used the composite emission factors shown in the table because the voyage data used in the model do not explicitly identify main engine speed ratings, i.e., slow or medium, or the auxiliary engine fuel type, i.e., marine distillate or residual marine. The composite factor for each pollutant is determined by weighting individual emission factors by vessel engine population data from a 2005 survey of ocean-going vessels that was performed by ARB.[60]

Table 2E-4 Main Engine Emission Factors by Ship and Fuel Type

Engine Type	**Main Engine Emission Factors (g/kW-hr)**						
	Fuel Type	**NO_X**	**PM_{10}**	**$PM_{2.5}$ [a]**	**HC**	**CO**	**SO_2**
Slow Speed	Residual Marine	18.1	1.5	1.4	0.6	1.4	10.5
Medium Speed	Residual Marine	14	1.5	1.4	0.5	1.1	11.5
Composite EF	Residual Marine	17.9	1.5	1.4	0.6	1.4	10.6

[a] Estimated from PM_{10} using a multiplicative adjustment factor of 0.92.

The emission factors for auxiliary engines are shown in Table 2E-5. The fuel specific main emission factors for NO_X and HC were taken from several recent analyses of ship emissions in the U.S., Canada, and Europe, as referenced above for the main engine load factors. The PM factor for marine distillate was taken from a report by ENVIRON International, which was also referenced above. The PM factor for residual marine was based on discussions with the California Air Resources Board (ARB) staff. The CO factors are from the Starcrest Consulting study of the Port of Los Angeles.[38] For SO_2, the fuel specific emission factors were obtained from Entec and Corbett and Koehler.'[56] The composite emission factors displayed in the table are discussed below.

Table 2E-5 Auxiliary Engine Emission Factors by Ship and Fuel Type

Engine Type	**Auxiliary Engine Emission Factors (g/kW-hr)**						
	Fuel Type	**NO_X**	**PM_{10}**	**$PM_{2.5}$ [a]**	**HC**	**CO**	**SO_2**
Medium Speed	Marine Distillate	13.9	0.3	0.3	0.4	1.1	4.3
Medium Speed	Residual Marine	14.7	1.5	1.4	0.4	1.1	12.3
Composite EF	Residual Marine	14.5	1.2	1.1	0.4	1.1	**

[a] Estimated from PM_{10} using a multiplicative adjustment factor of 0.92.
[b] See Table 2E-6 for composite SO_2 emission factors by vessel type.

As for main engines, the STEEM study used the composite emission factors for auxiliary engines. For all pollutants other than SO_2, underlying data used in the model do not explicitly identify auxiliary engine voyages by fuel type, i.e., marine distillate or residual marine. Again, the composite factor for those pollutants was determined by weighting individual emission factors by vessel engine population data from a 2005 survey of ocean-going vessels that was performed by ARB.[61]

For SO_2, composite emission factors for auxiliary engines were calculated for each vessel type. These composite factors were determined by taking the fuel specific emission factors from Table 2E-5 and weighting them with an estimate of the amount of marine distillate and residual marine that is used by these engines. The relative amount of each fuel type consumed was taken from the 2005 ARB survey. The relative amounts of each fuel type for each vessel type and the resulting SO_2 emission factors are shown in Table 2E-6.

Table 2E-6 Auxiliary Engine SO_2 Composite Emission Factors by Vessel Type

Vessel Type	Residual Marine (%)	Marine Distillate (%)	*Composite Emission Factor (g/kW-hr)*
Bulk Carrier	71	29	*9.98*
Container Ship	71	29	*9.98*
General Cargo	71	29	*9.98*
Passenger Ship	92	8	*11.66*
Refrigerated Cargo	71	29	*9.98*
Roll On-Roll Off	71	29	*9.98*
Tanker	71	29	*9.98*
Miscellaneous	*0*	*100*	*4.3*

2.6.7 Adjustments to STEEM PM and SO_2 Emission Inventories

The interport emission results contained in this study for PM_{10} and SO_2 were taken from the STEEM inventories and then adjusted to reflect the U.S. Government's recent review of available engine test data and fuel sulfur levels for the near port analysis. In the near ports work, a PM emission factor of 1.4 g/kW-hr was used for most main engines, e.g., slow speed diesel and medium speed diesel engines, all of which are assumed to use residual marine. A slightly higher value was used for steam turbine and gas turbine engines, and a slightly lower value was used for most auxiliary engines. However, these engines represent only a small fraction of the total emissions inventory. As shown in Section 2.6.6, the STEEM study used an emission factor of 1.5 g/kW-hr for all main engines and a slightly lower value for auxiliary engines. Here again, the auxiliary engines comprise only a small fraction of the total emissions from these ships. Therefore, for simplicity, the interport PM inventories were adjusted by multiplying the STEEM results by the ratio of the two primary emission factors, i.e., 1.4/1.5 or 0.933, to approximate the difference in fuel effects.

Appendix 2F: Inventories Used for Air Quality Modeling

The emission inventories presented in this chapter are slightly different from the emissions inventories used in the air quality modeling presented in Chapter 3. Specifically, the inventories used in the air quality modeling reflect a slightly different boundary for the proposed ECA that was based on a measurement error. Due to the nature of the measurement error, the corrections to the ECA boundaries are not uniform, but are different by coastal area. As seen in Table 2F-1, the changes are not expected to have a significant impact on the results of our analysis. The measurement error affects only those portions that are farthest from shore.

The inventories used for air quality modeling also only contain Tier I NO_X controls, as opposed to the Tier I and Tier II controls contained in the final inventories.

A comparison of the air quality and final inventories by region for the 2020 baseline scenarios is provided in Table 2F-1. Results are provided only for NO_X, $PM_{2.5}$, and SO_2, since the air quality modeling is focused on ozone and $PM_{2.5}$. As shown, the inventory provided for air quality modeling generally understates the inventory reductions and air quality benefits produced by the ECA.

Table 2F-1 Comparison of Air Quality Inventories vs Final Inventories for 2020 Baseline Case

U.S. Region	Metric Tonnes per Year								
	NO_X			$PM_{2.5}$			SO_X		
	AQ	Final	% Diff	AQ	Final	% Diff	AQ	Final	% Diff
East Coast (EC)	439,713	391,995	12%	35,891	35,882	0%	323,108	323,038	0%
Gulf Coast (GC)	261,024	232,114	12%	21,669	21,531	1%	175,862	174,751	1%
North Pacific (NP)	42,291	38,051	11%	3,575	3,603	-1%	27,580	27,807	-1%
South Pacific (SP)	216,849	208,294	4%	17,092	18,536	-8%	138,102	149,751	-8%
Great Lakes (GL)	19,842	18,768	6%	1,484	1,484	0%	11,993	11,993	0%
Total 48-State	**979,719**	**889,222**	10%	**79,711**	**81,037**	-2%	**676,645**	**687,339**	-2%

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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 $PM_{10\text{-}2.5}$). 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 (≥ 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 µg/m^3 and 0.2 µg/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 μg/m^3 with a few areas having higher levels and some areas in the west having lower levels.

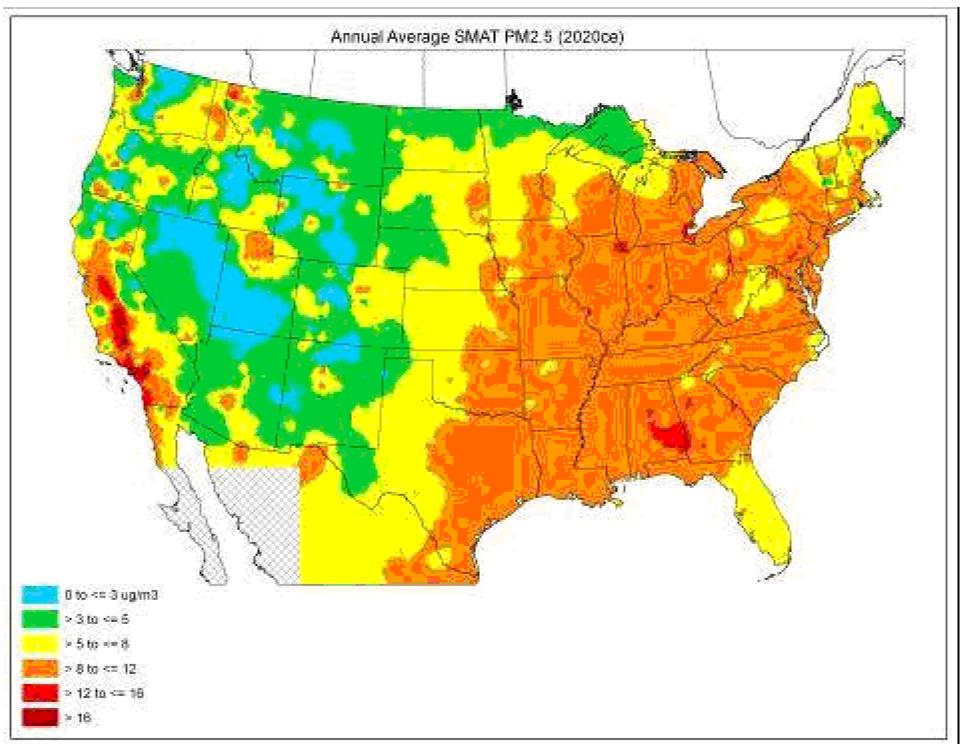

Figure 3.2-1 Annual Average $PM_{2.5}$ Concentrations in 2020 without an ECA

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

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[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 $\mu g/m^3$.

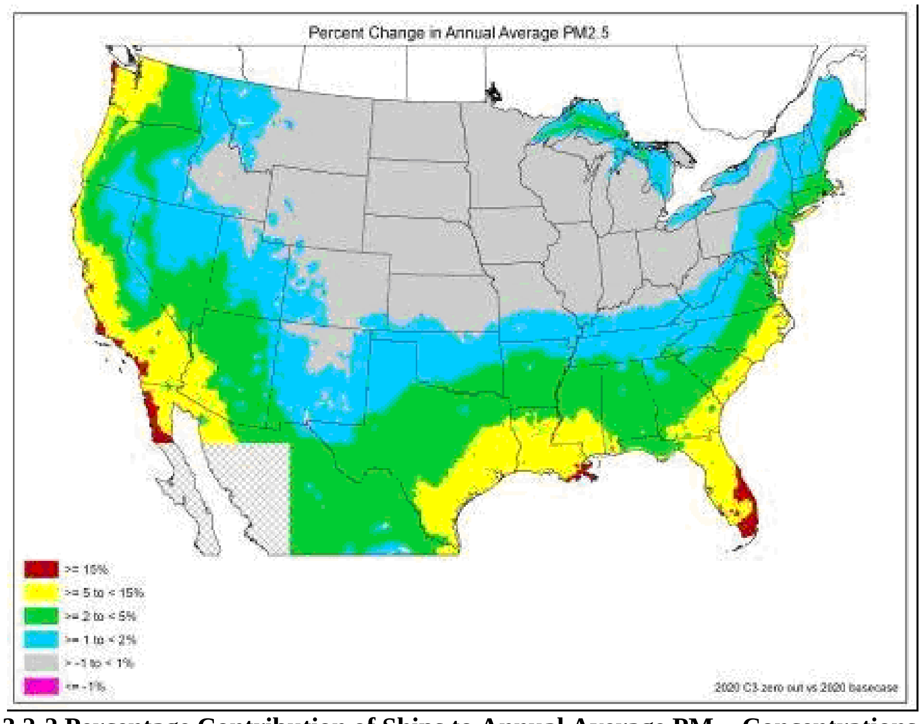

Figure 3.2-2 Percentage Contribution of Ships to Annual Average $PM_{2.5}$ Concentrations in 2020

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

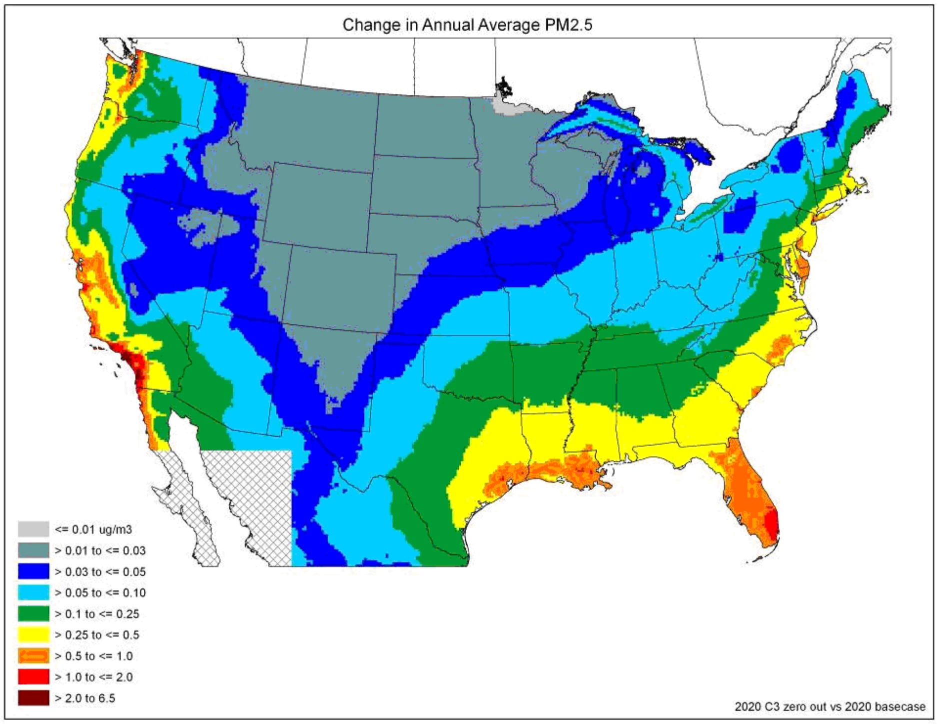

Figure 3.2-3 Absolute Contribution of Ships to Annual Average $PM_{2.5}$ Concentrations in 2020

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.

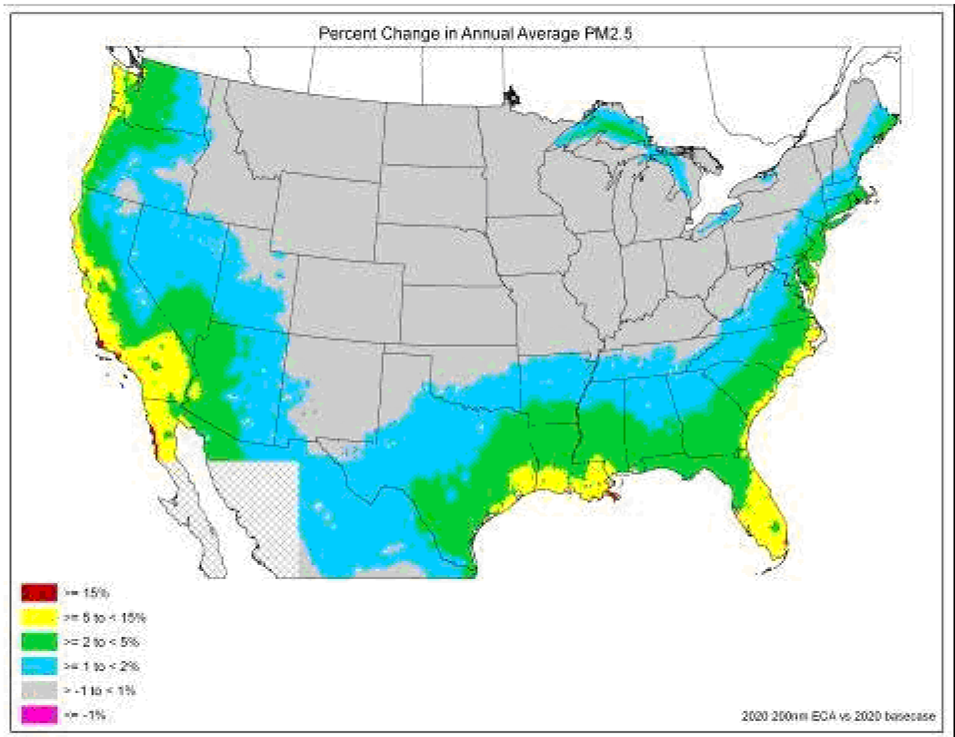

Figure 3.2-4 Percent Improvement in Annual Average $PM_{2.5}$ Concentrations in 2020 Resulting from the Application of the Proposed ECA

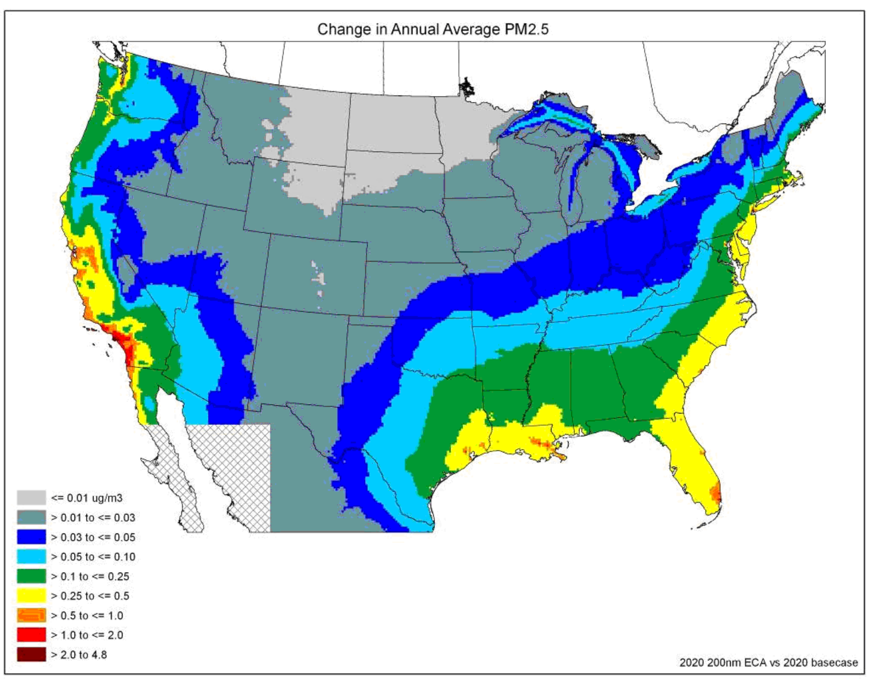

Figure 3.2-5 Absolute Improvement in Annual Average $PM_{2.5}$ Concentrations in 2020 Resulting from the Application of the Proposed ECA

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.

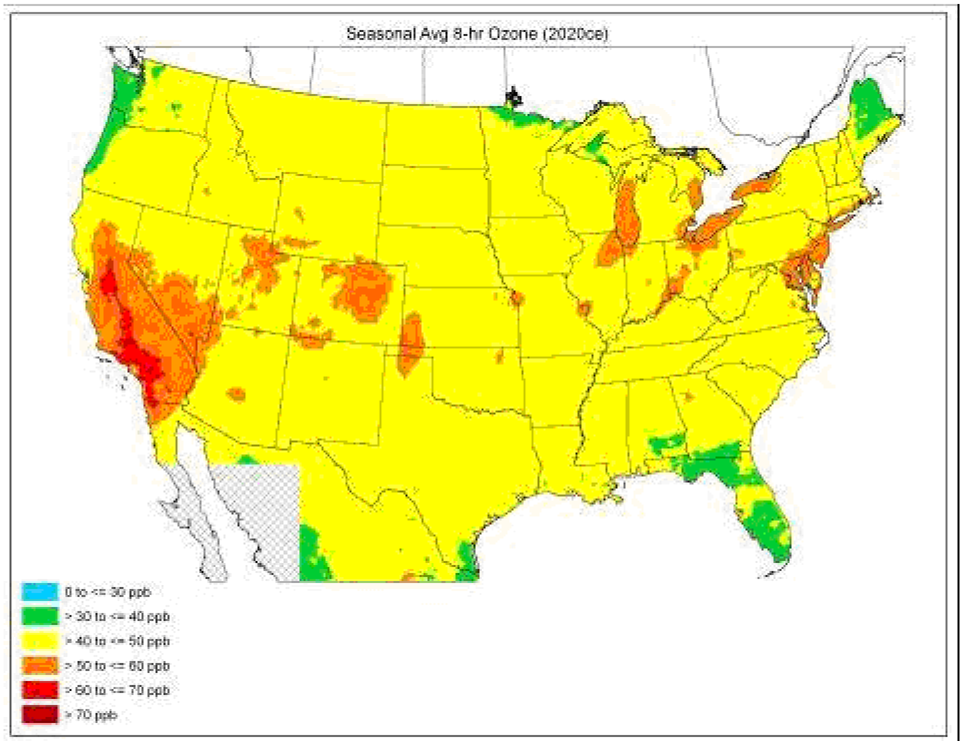

Figure 3.2-6 Seasonal Average of Daily 8-hour Maximum Ozone Concentrations in 2020 without an ECA

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.

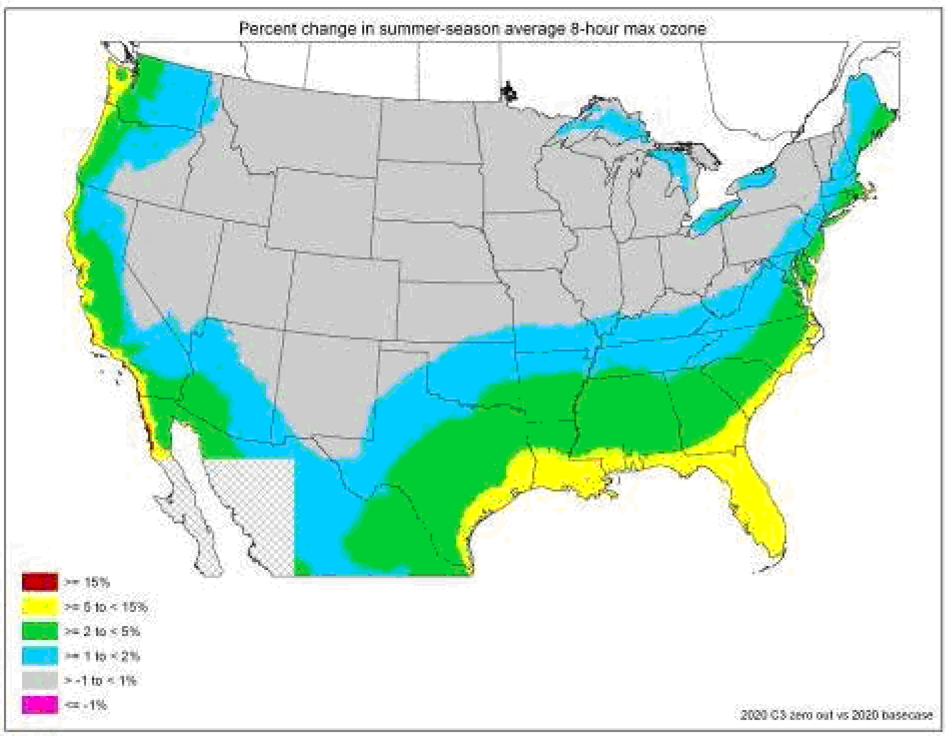

Figure 3.2-7 Percentage Contribution of Ships to Summertime Maximum 8-hour Average Ozone Concentrations in 2020

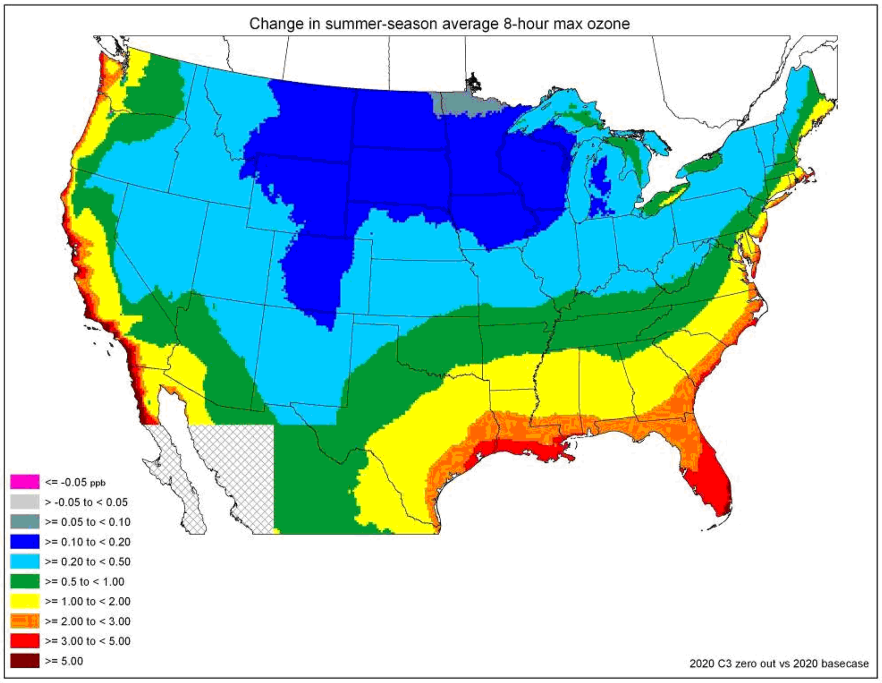

Figure 3.2-8 Absolute Contribution of Ships to Summertime Maximum 8-hour Average Ozone Concentrations in 2020

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.

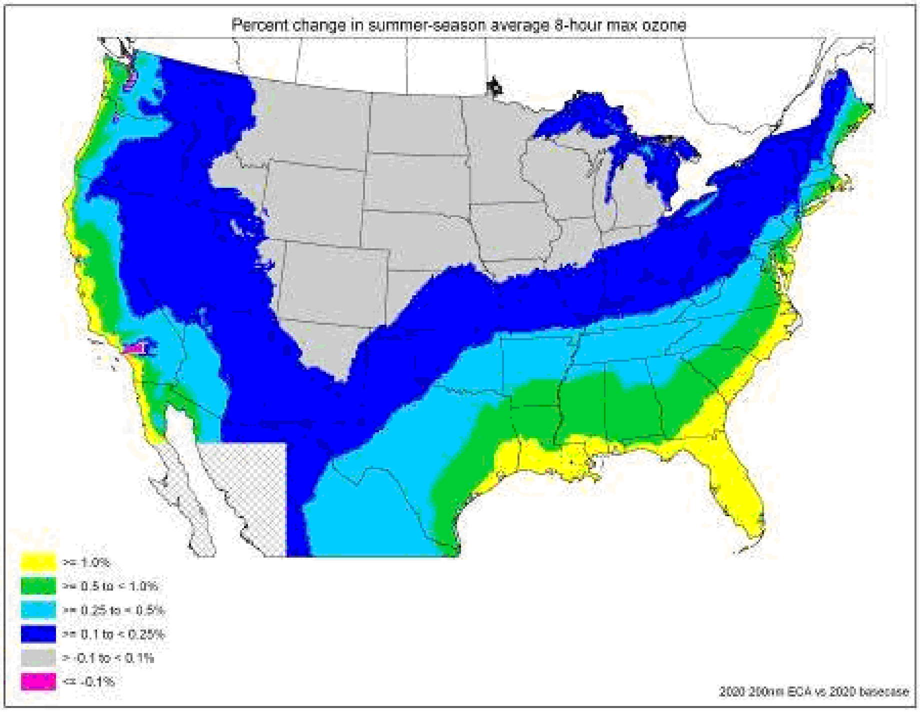

Figure 3.2-9 Percent Improvement in Summertime Maximum 8-hour Average Ozone Concentrations in 2020 Resulting from the Application of the Proposed ECA

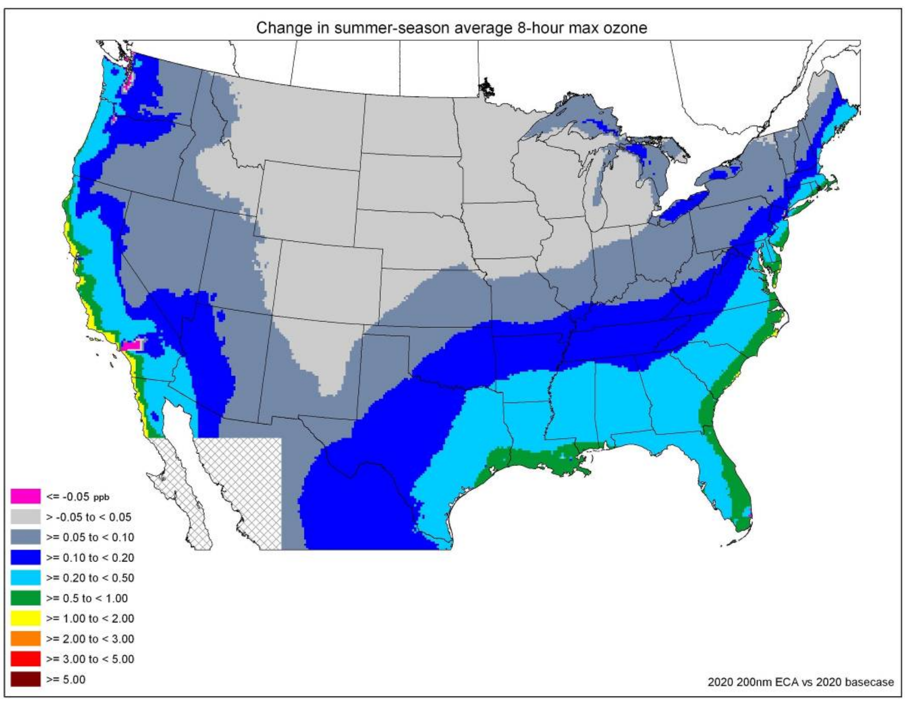

Figure 3.2-10 Absolute Improvement in Summertime Maximum 8-hour Average Ozone Concentrations in 2020 Resulting from the Application of the Proposed ECA

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

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[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.

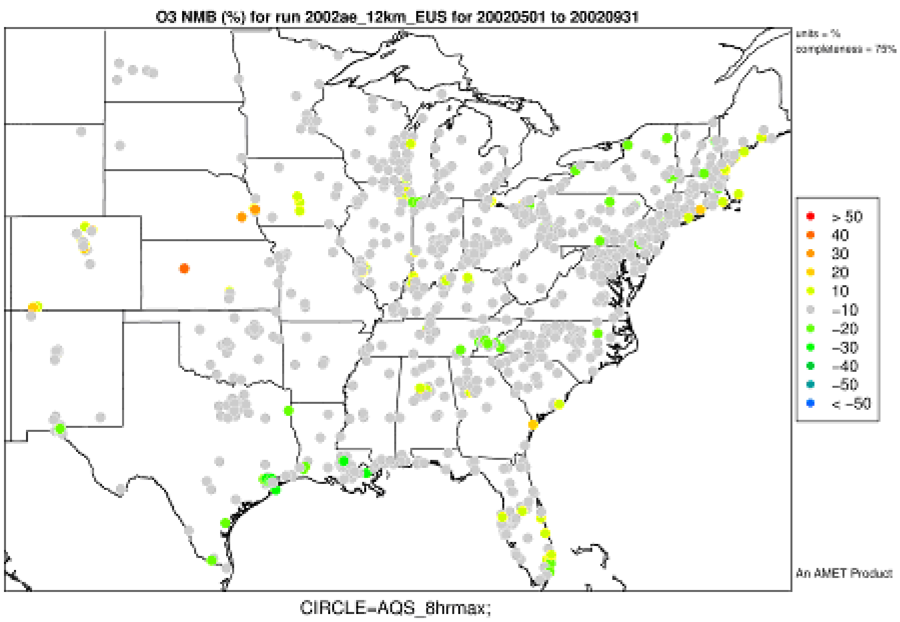


Figure 3.2-12. Normalized Mean Bias (%) of hourly ozone (40 ppb threshold) by monitor for 12-km Eastern U.S. domain, seasonal aggregate

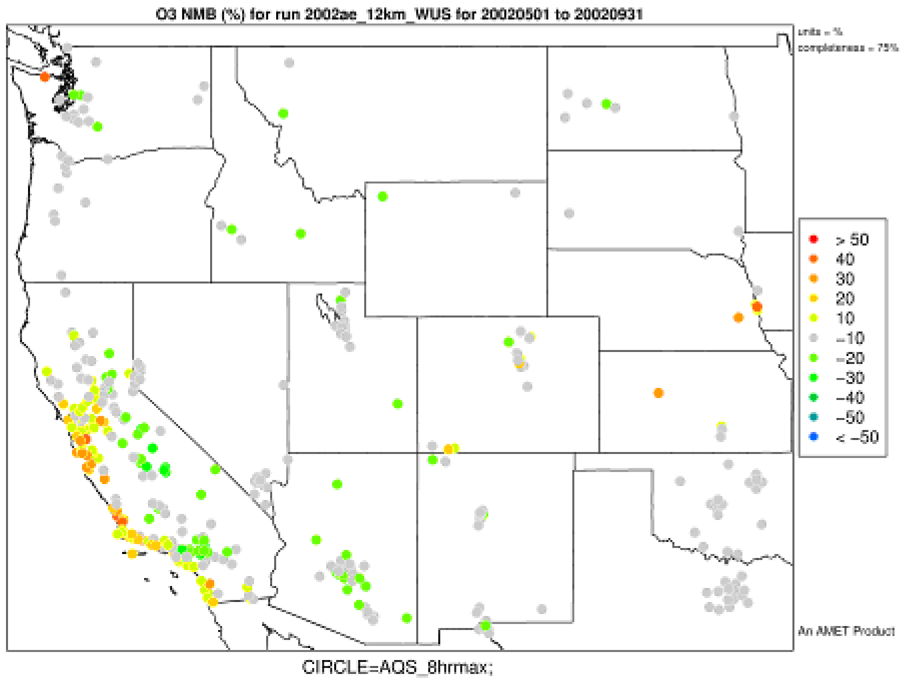


Figure 3.2-13. Normalized Mean Bias (%) of hourly ozone (40 ppb threshold) by monitor for 12-km Western U.S. domain, seasonal aggregate.

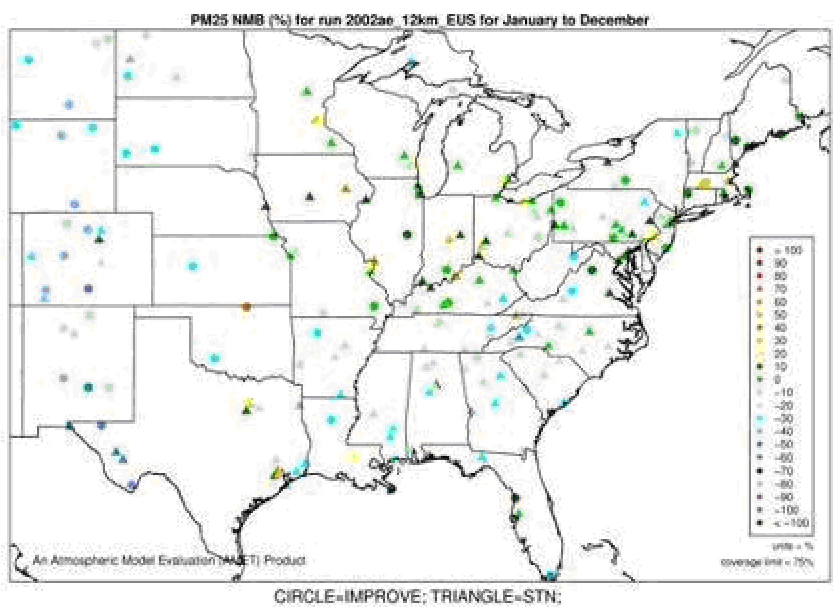


Figure 3.2-14. Normalized Mean Bias (%) of annual $PM_{2.5}$ by monitor for 12-km Eastern U.S. domain, 2002

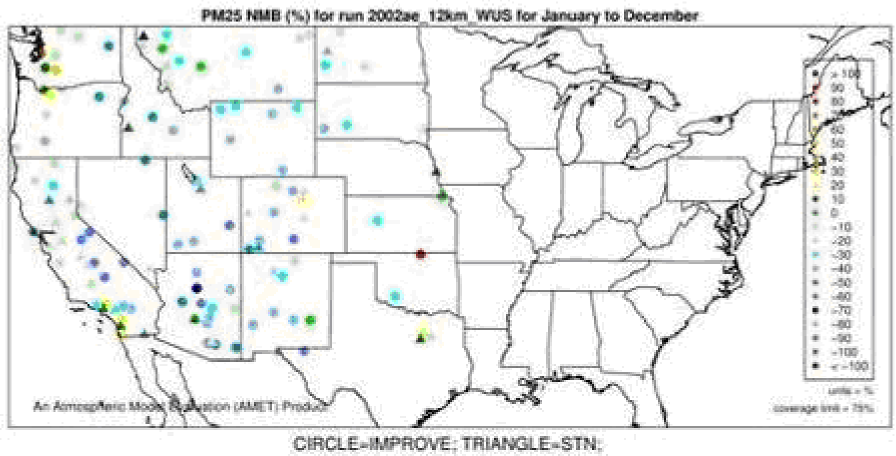


Figure 3.2-15. Normalized Mean Bias (%) of annual PM2.5 by monitor for 12-km Western U.S. domain, 2002

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.

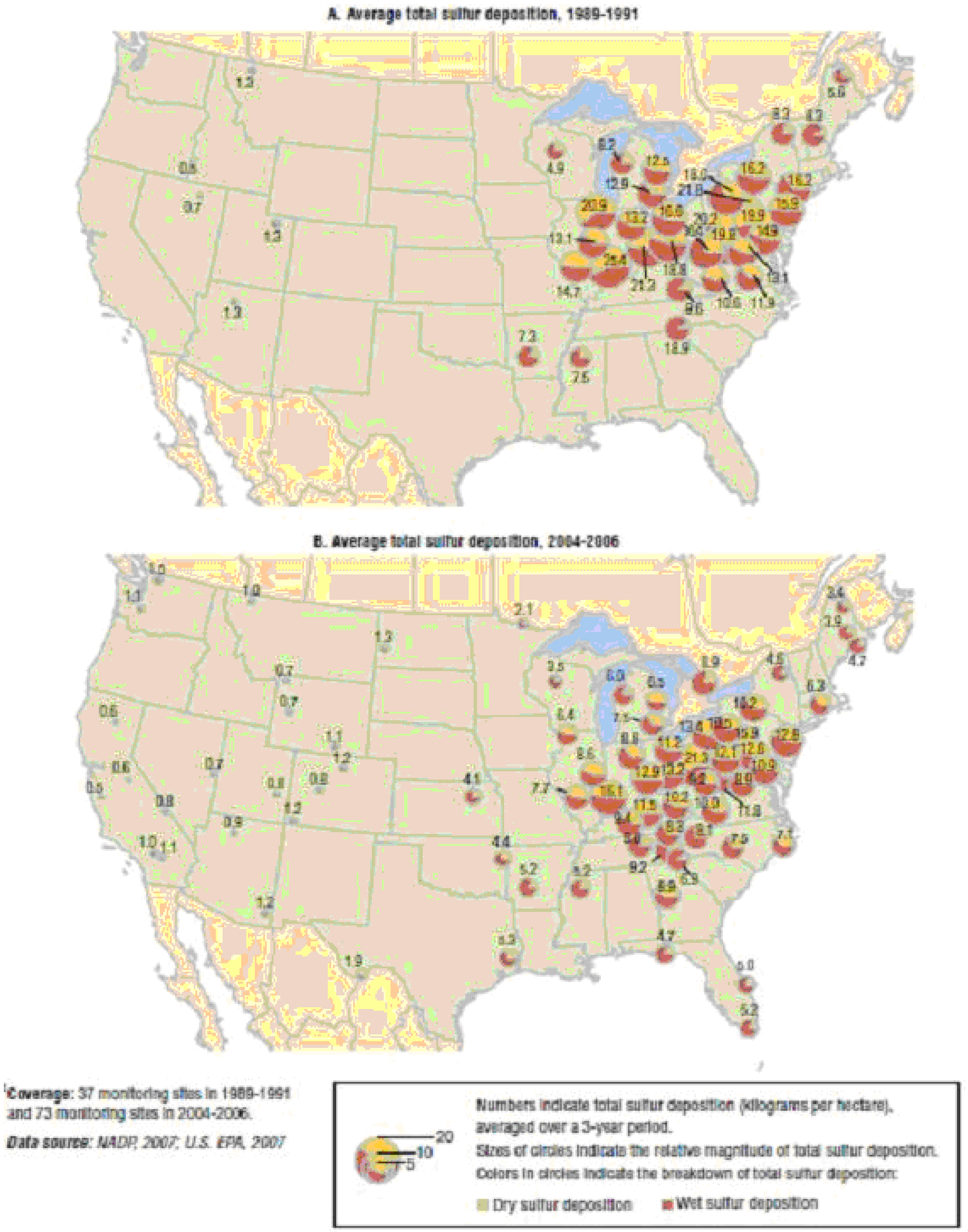


Figure 3.3-1 Total Sulfur Deposition in the Contiguous U.S., 1989-1991 and 2004 -2006

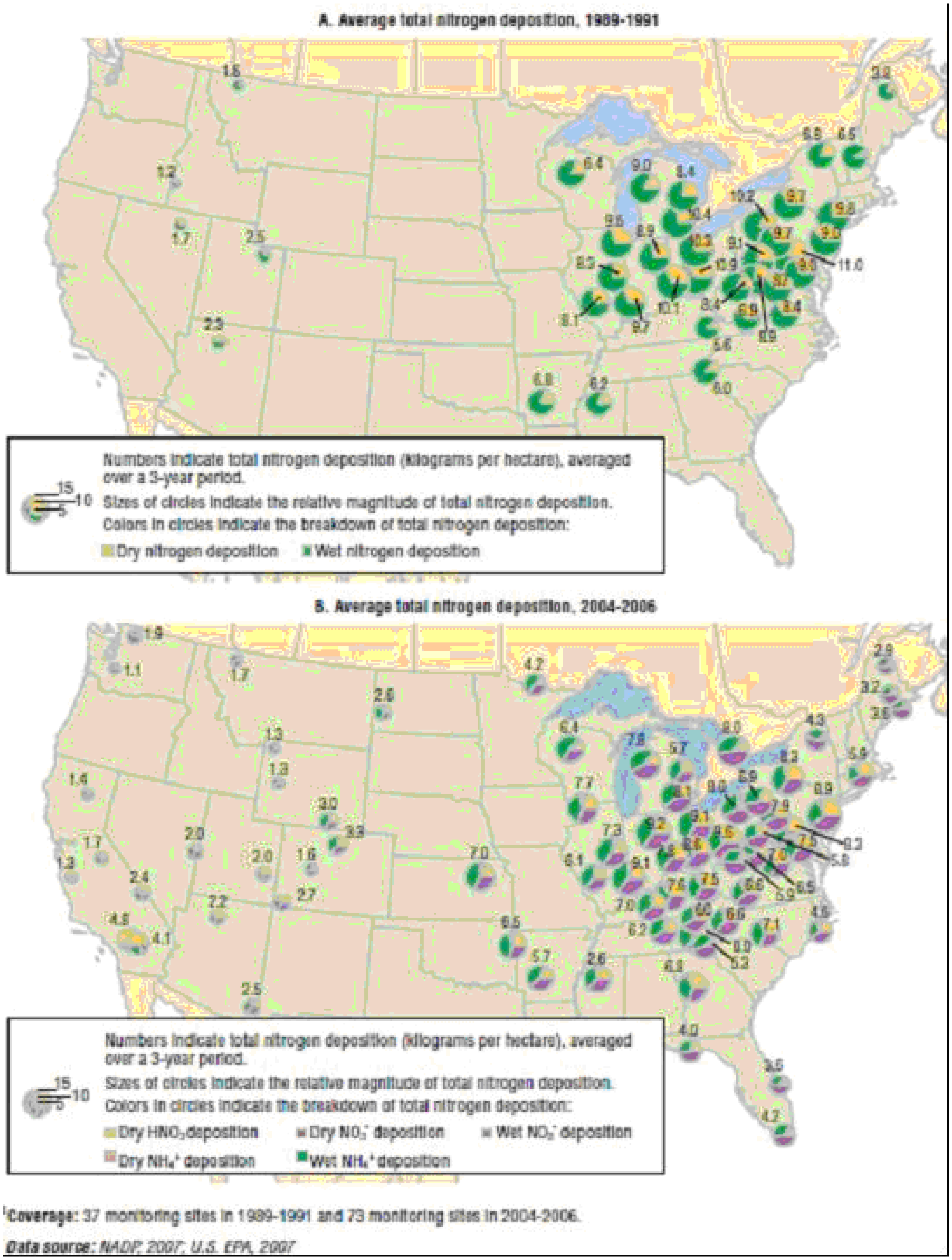


Figure 3.3-2 Total Nitrogen Deposition in the Contiguous U.S., 1989-1991 and 2004-2006

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

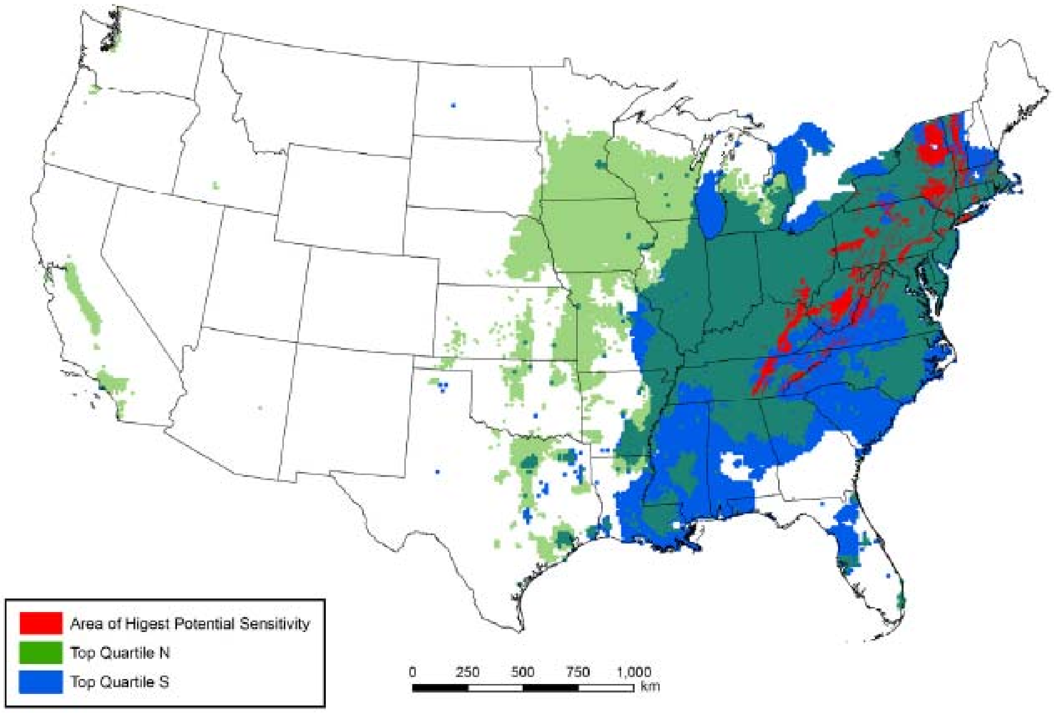


Figure 3.3-3 Areas Potentially Sensitive to Terrestrial Acidification

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]

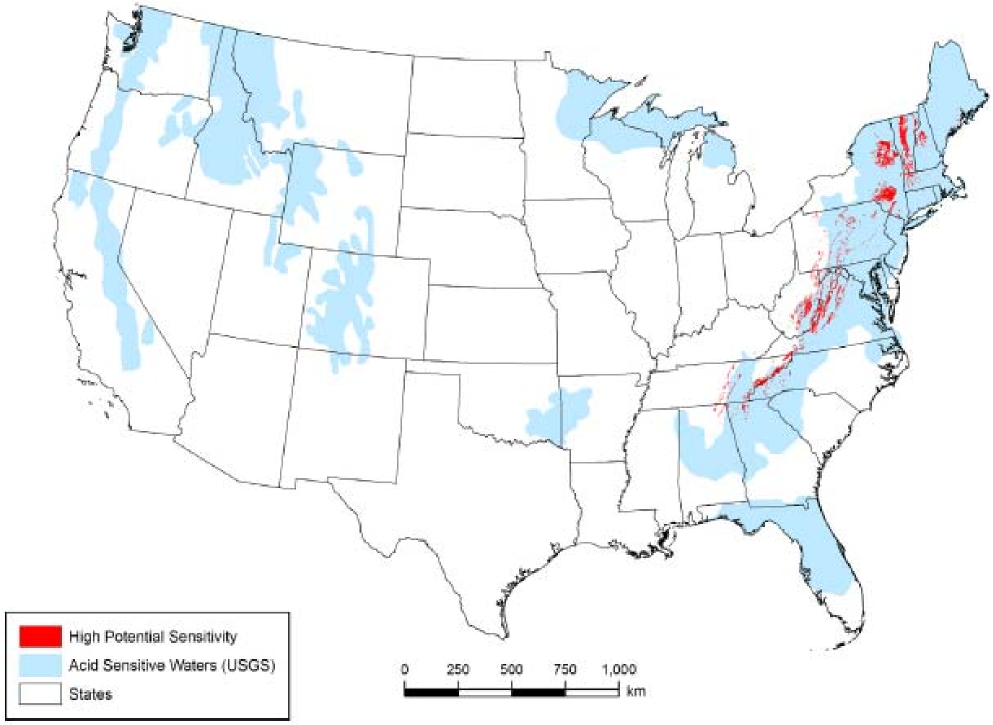


Figure 3.3-4 Areas Potentially Sensitive to Aquatic Acidification

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]

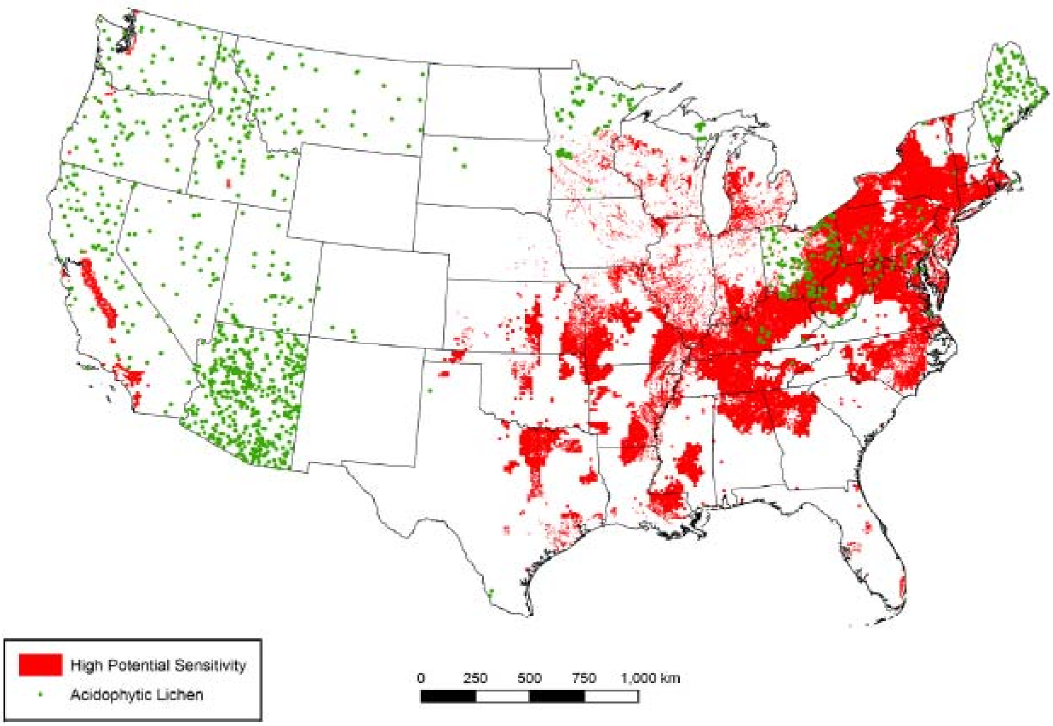


Figure 3.3-5 Areas Potentially Sensitive to Terrestrial Nutrient Enrichment

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]

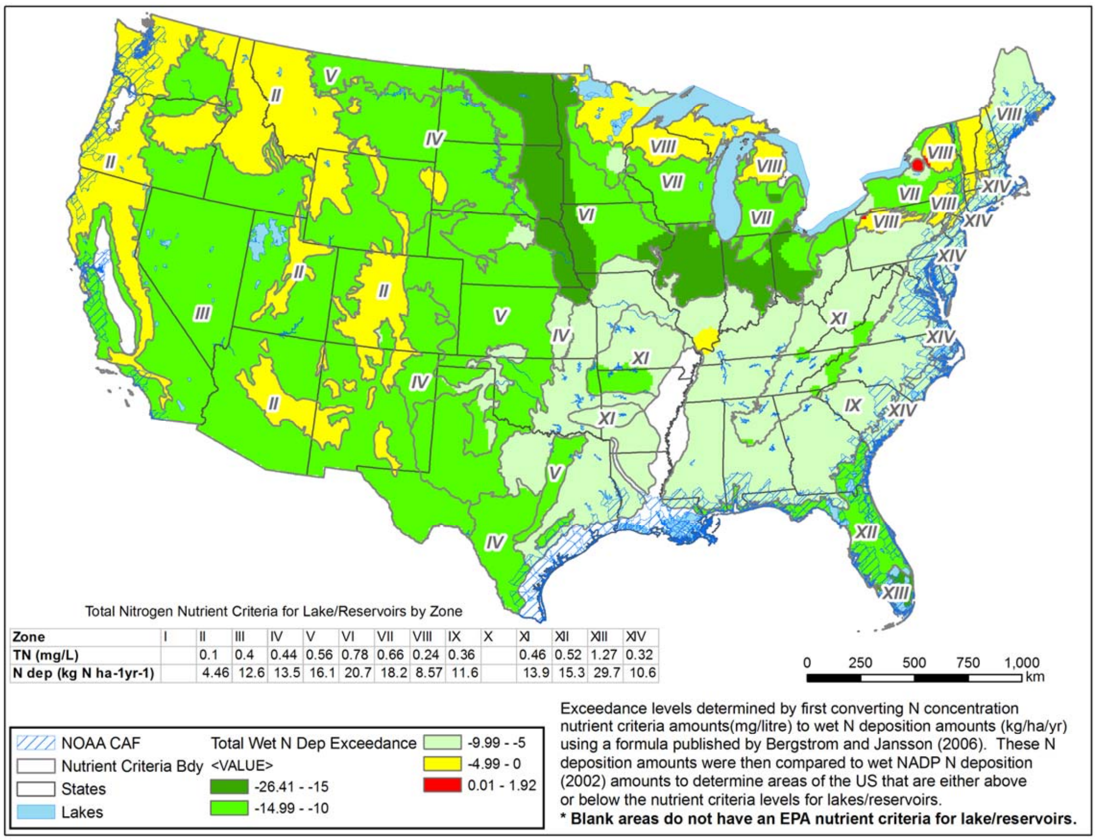
Nitrogen deposition is the main source of nitrogen for many surface waters in the U.S. including headwater streams, lower order streams, and high elevation lakes.[182,183] Elevated surface water nitrate concentrations due to nitrogen deposition occur in both the eastern and western U.S. although high 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.



Zone	I	II	III	IV	V	VI	VII	VIII	IX	X	XI	XII	XIII	XIV
TN (mg/L)		0.1	0.4	0.44	0.56	0.78	0.66	0.24	0.36		0.46	0.52	1.27	0.32
N dep (kg N ha-1yr-1)		4.46	12.6	13.5	16.1	20.7	18.2	8.57	11.6		13.9	15.3	29.7	10.6

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.

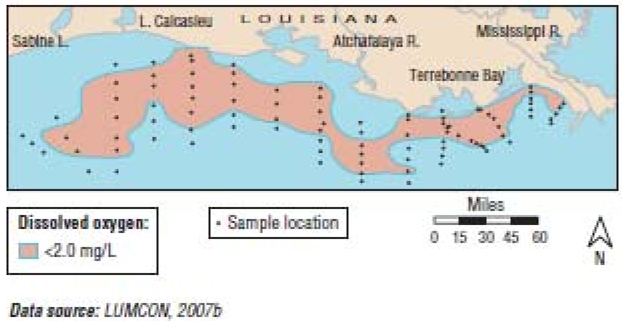


Figure 3.3-7 a. Hypoxia Zone in 2007 for the Gulf of Mexico

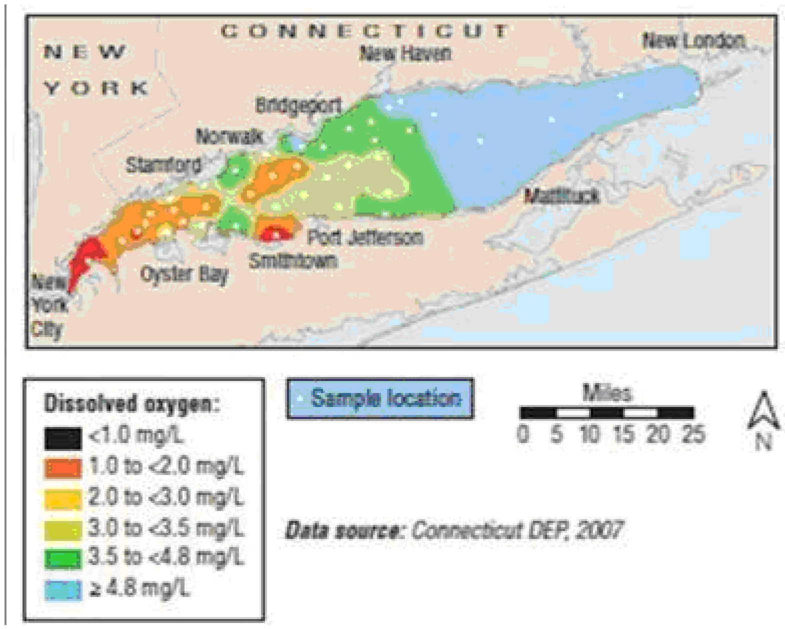
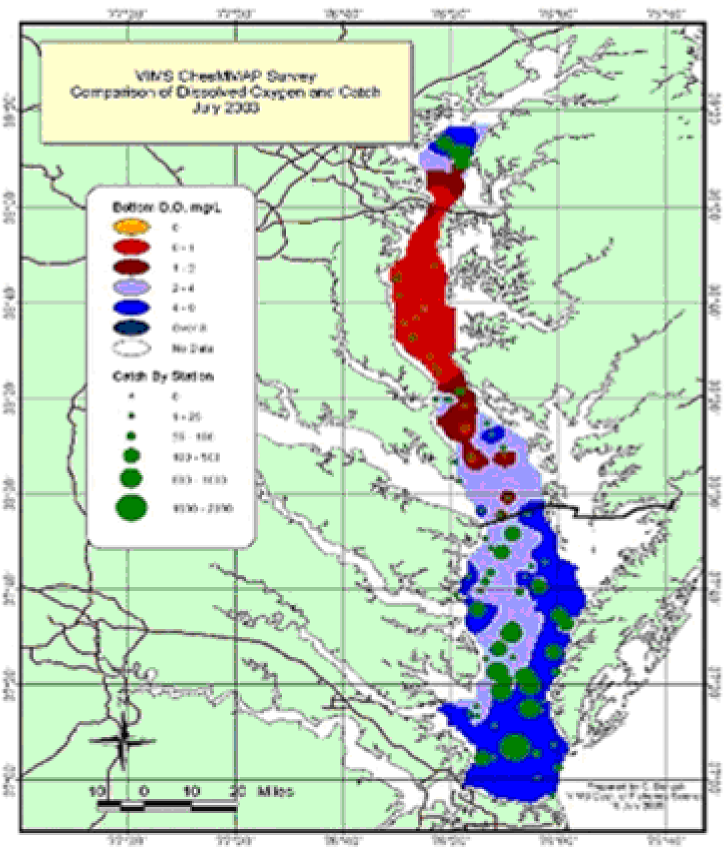


Figure 3.3-7 b. Hypoxia Zone in 2007 for Long Island Sound



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.

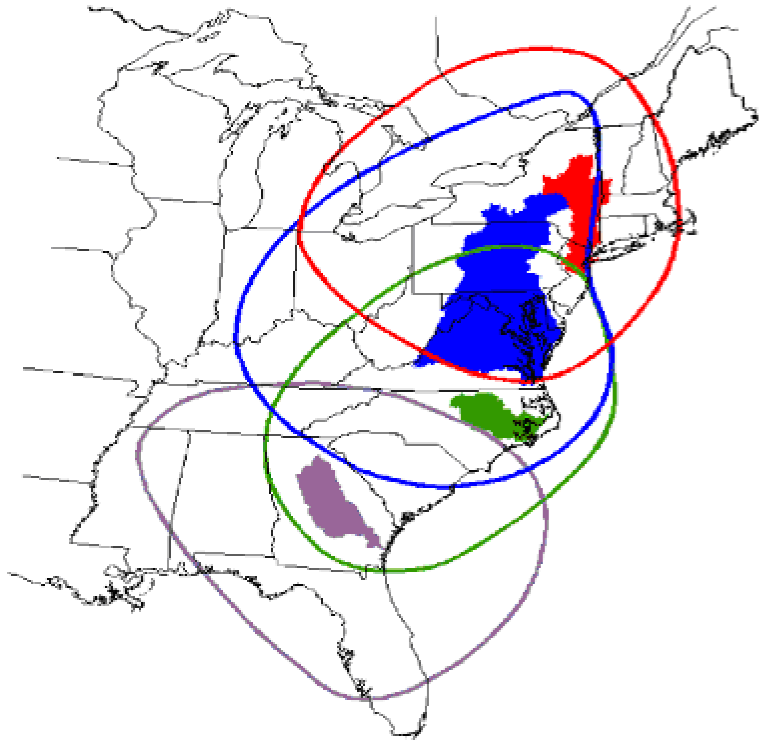


Figure 3.3-8 Principal Airsheds and Watersheds for Oxides of Nitrogen for Estuaries.Hudson/Raritan Bay; Chesapeake Bay; Pamlico Sound; and Altamaha Sound (listed from north to south).

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.

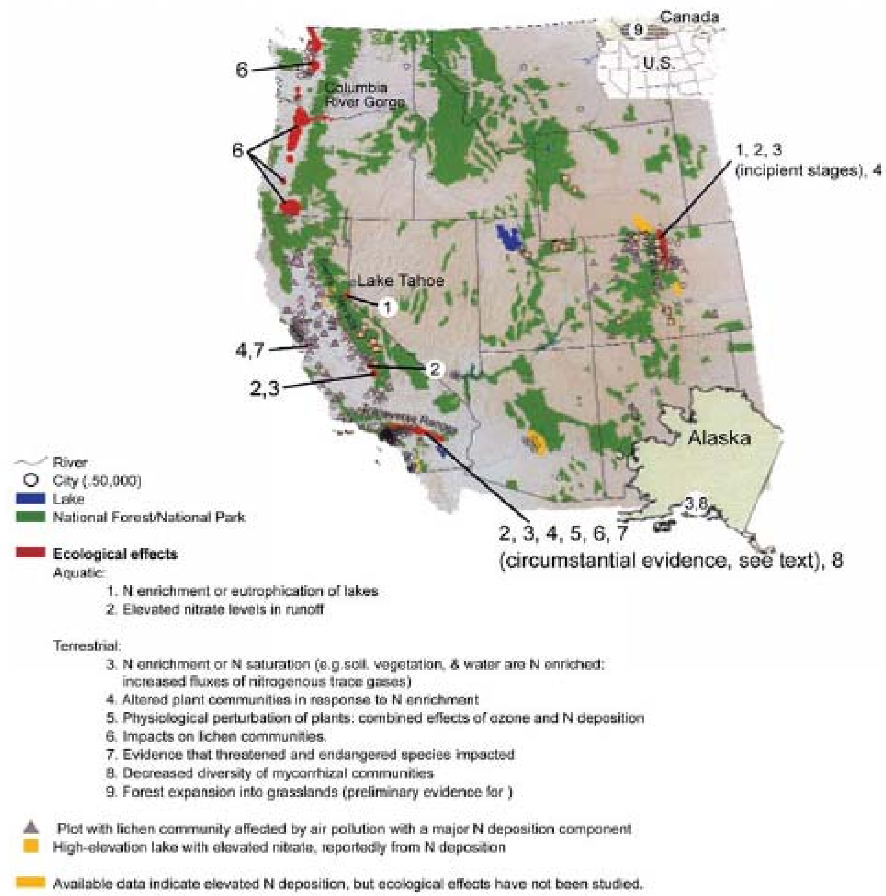
Although there has been no direct evidence of a physiological association between tree injury and heavy metal exposures, heavy metals have been implicated because of similarities between metal deposition patterns and forest decline.[221] This hypothesized correlation was further explored in high elevation 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.



Source: (Fenn et al., 2003a). Reprinted with permission.

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,

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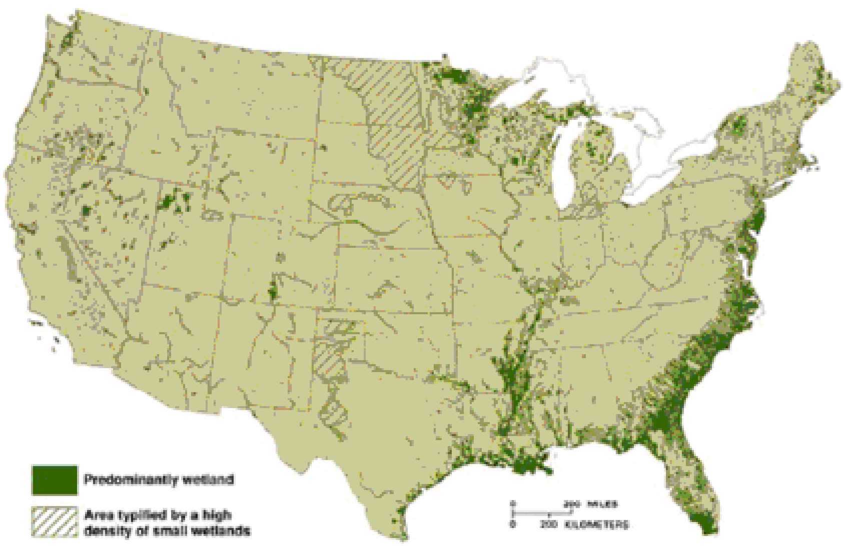
[Y] In Stage 0, nitrogen inputs are low and there are strong nitrogen limitations on growth. Stage 1 is characterized by high nitrogen rentention and fertilization effect of added nitrogen on tree growth. Stage 2 includes the induction of nitrification and some nitrate leaching, though growth may still 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]



Source: Data were obtained from the National Land Cover Data (NLCD) (2001) (http://www.mrlc.gov/)

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 taxonomic 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).

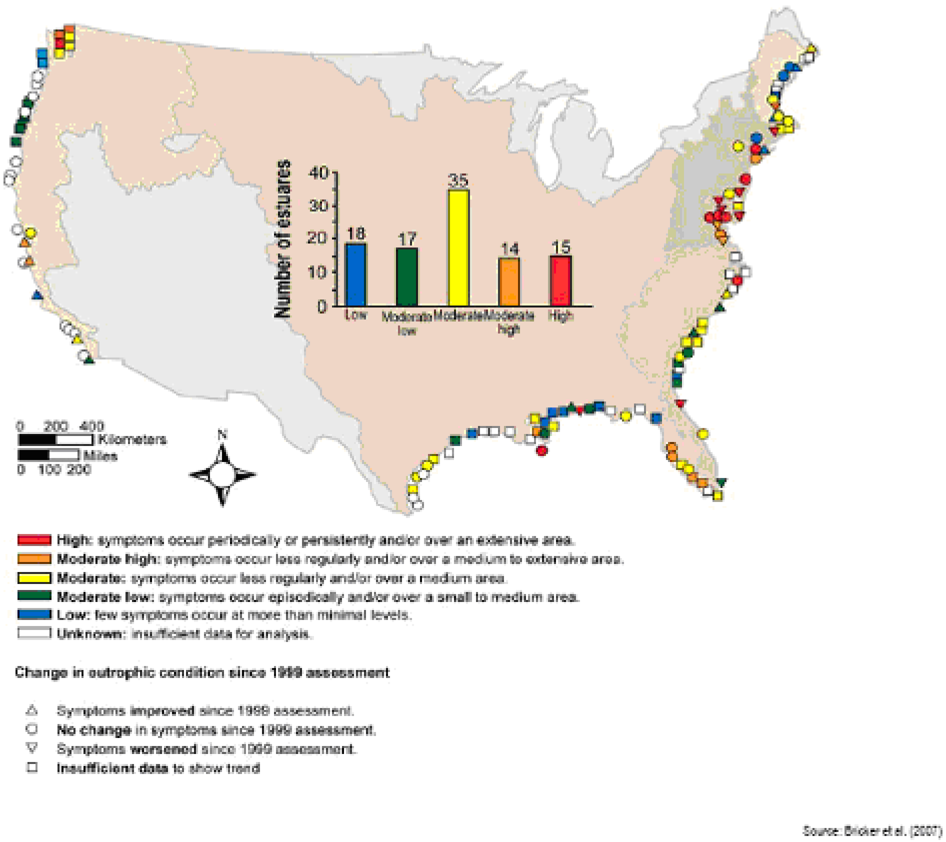


Figure 3.3-11 Overall Eutrophication Condition on a National Scale

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.

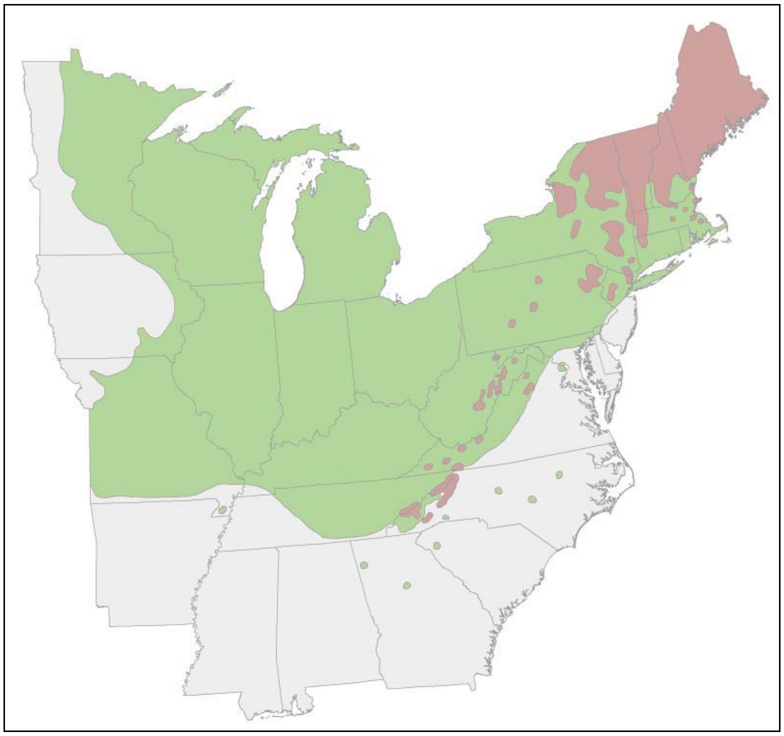
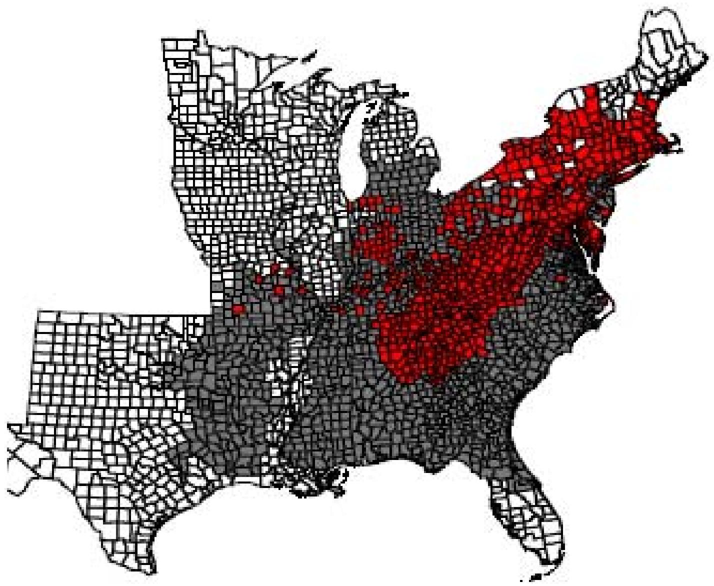


Figure 3.3-12 Distribution of Red Spruce (pink) and Sugar Maple (green) in the Eastern U.S.[297]

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: Holzmueler 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.

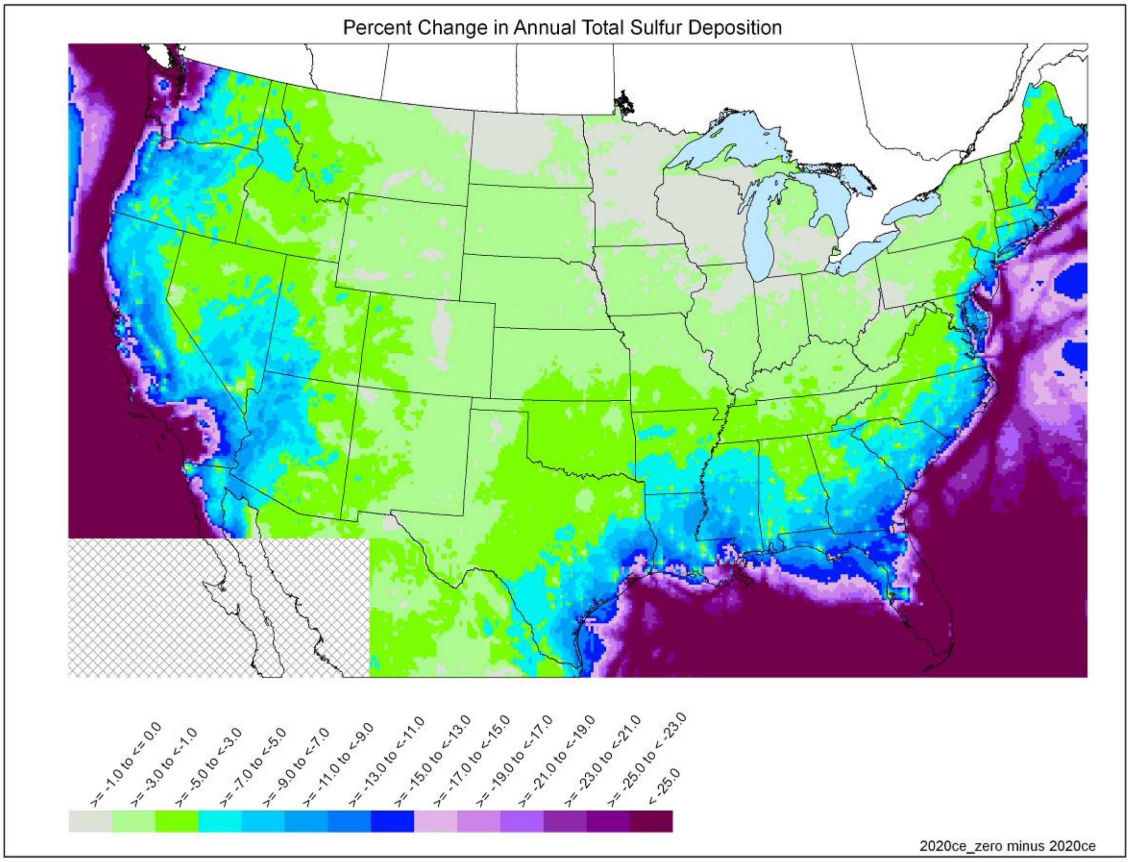


Figure 3.3-14 Percent Contribution in 2020 of Ships to Annual Total Sulfur Deposition in the U.S.

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).

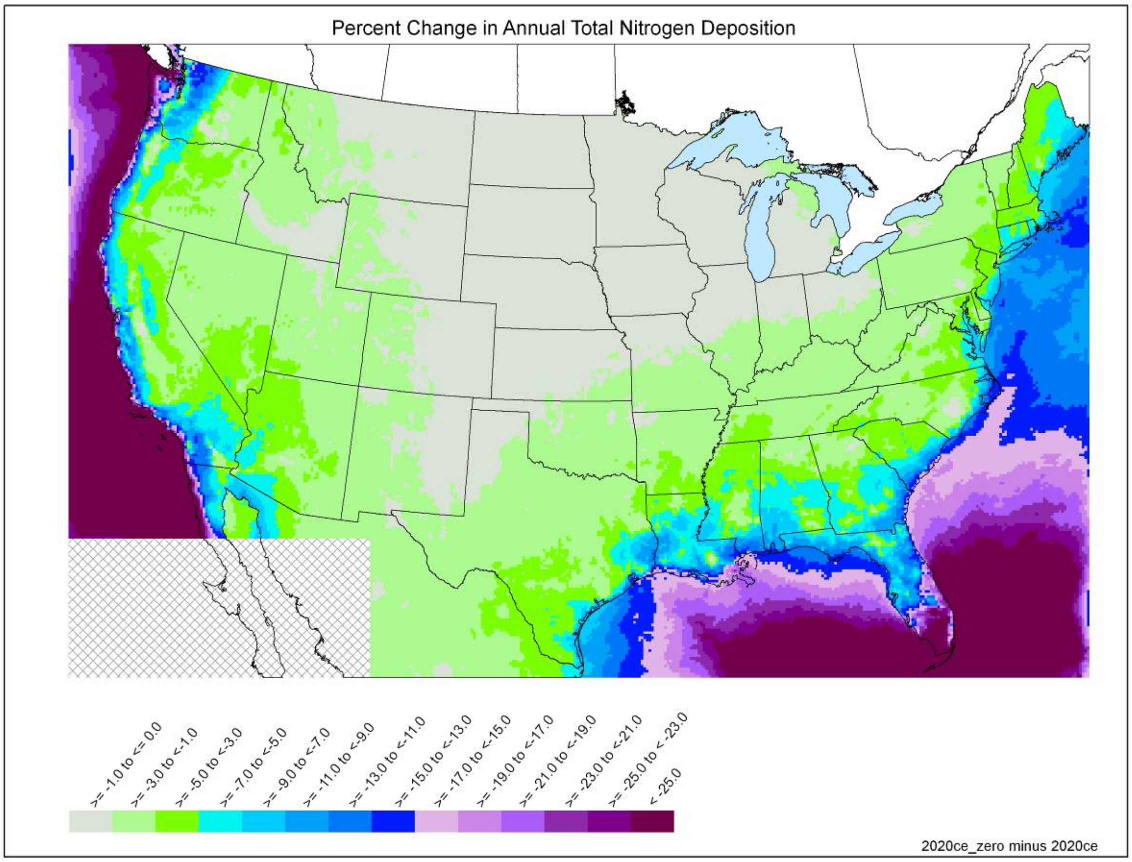


Figure 3.3-15 Percent Contribution in 2020 of Ships to Annual Total Nitrogen Deposition in the U.S.

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.

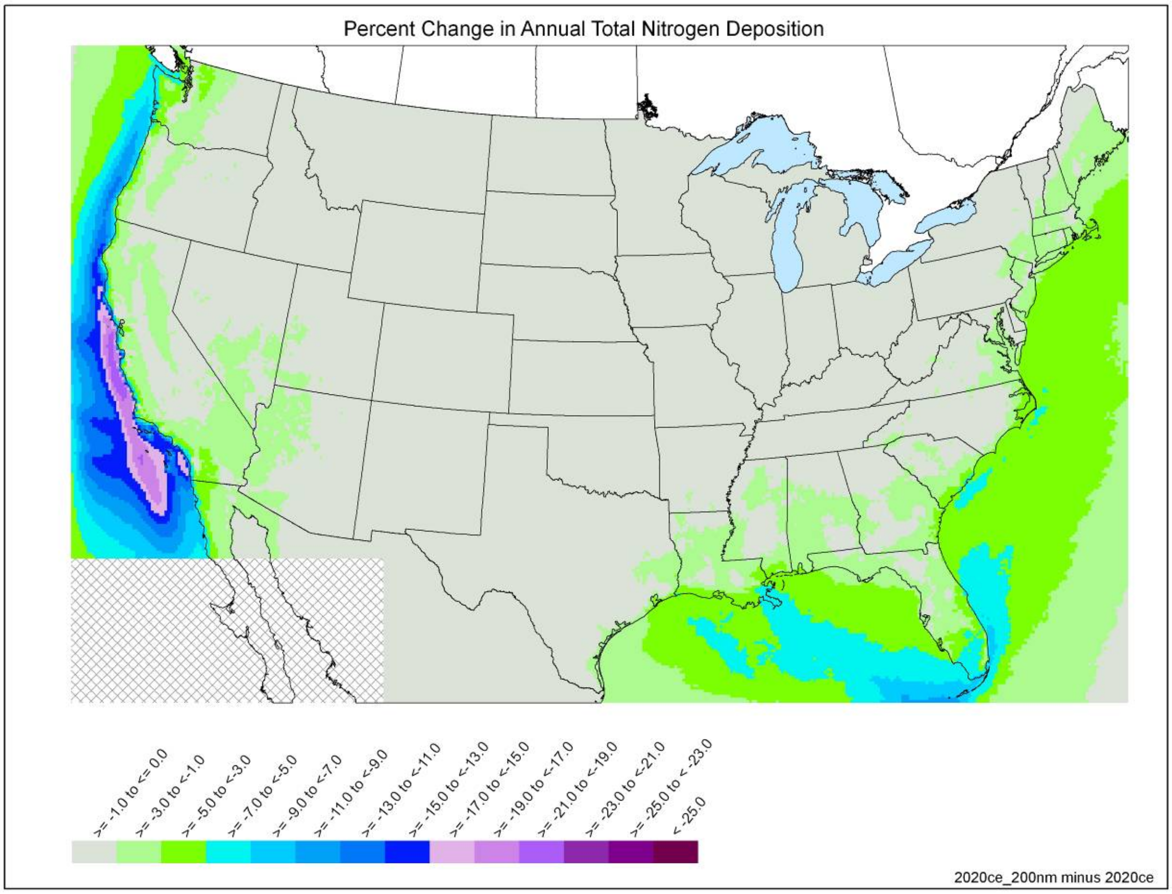


Figure 3.3-16 Percent Change in Annual Total Nitrogen over the U.S. Modeling Domain for the ECA Modeling Scenario.

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.

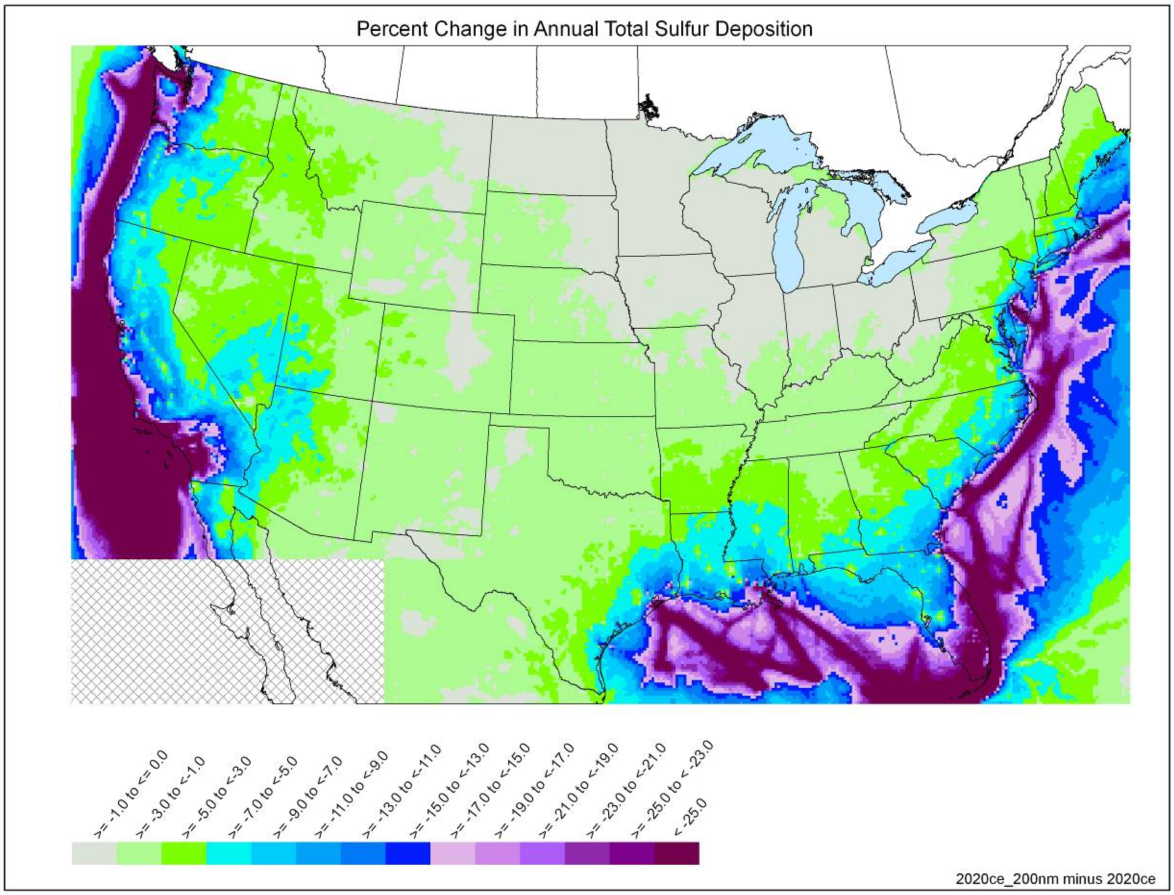


Figure 3.3-17 Percent Change in Annual Total Sulfur over the U.S. Modeling Domain for the ECA Modeling Scenario.

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.

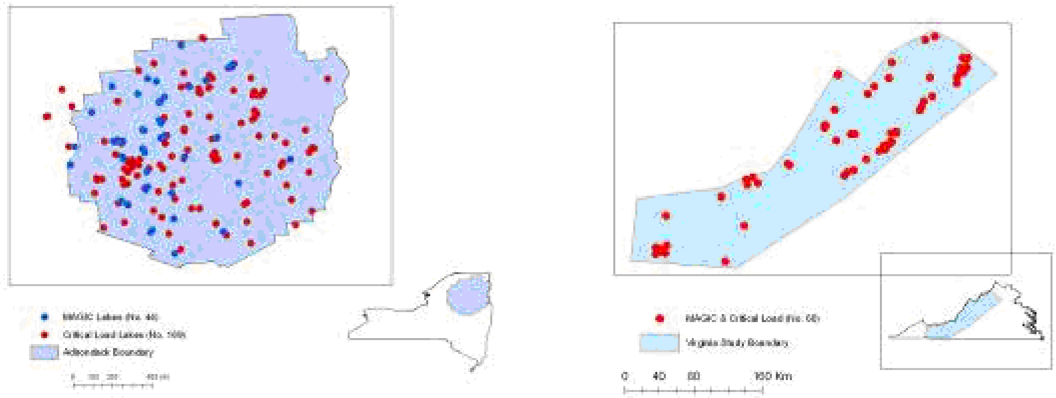


Figure 3.3-18 Locations of lakes and streams used in this assessment

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

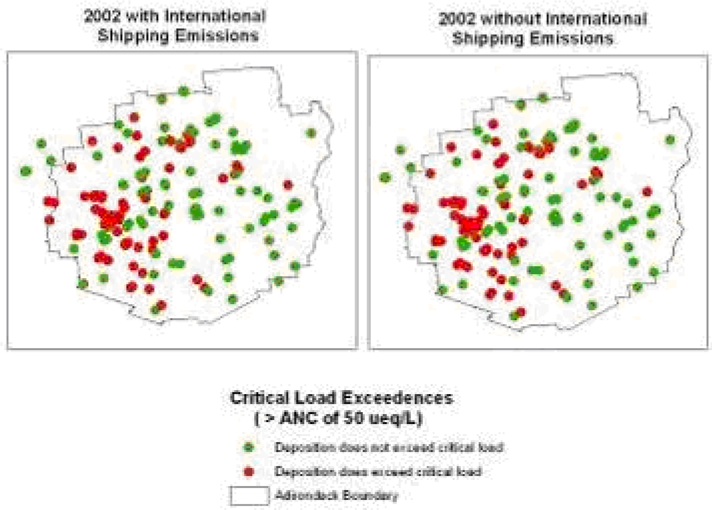


Figure 3.3-19 a. 2002

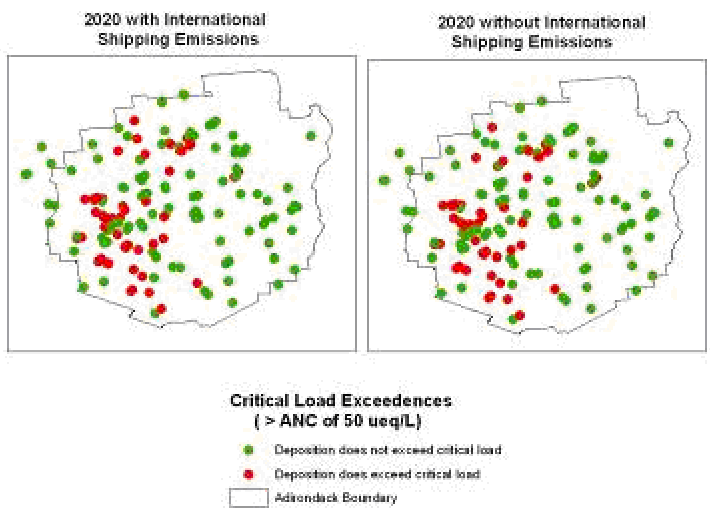


Figure 3.3-19 b. 2020; Critical Load Exceedance for ANC Concentration of 50 µeq/L. Green dots represent lakes in the Adirondack Mountains where current nitrogen and sulfur deposition is below their critical load and maintains an ANC concentration of 50 µeq/L. Red dots are lakes where current nitrogen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

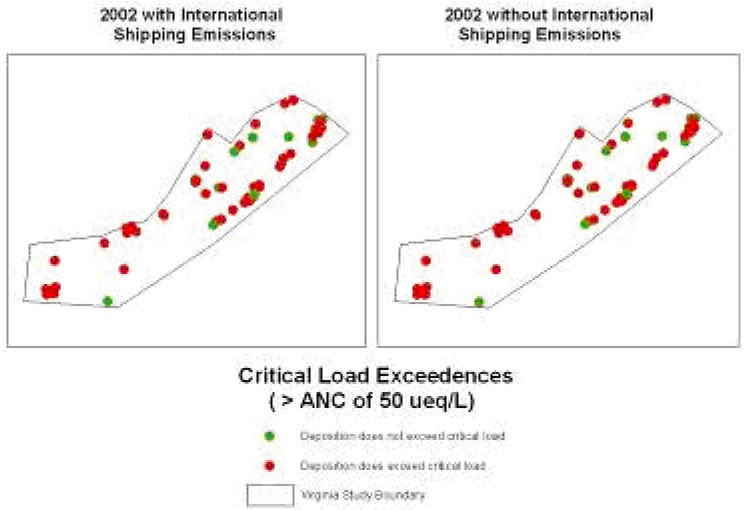


Figure 3.3-20 a. 2002

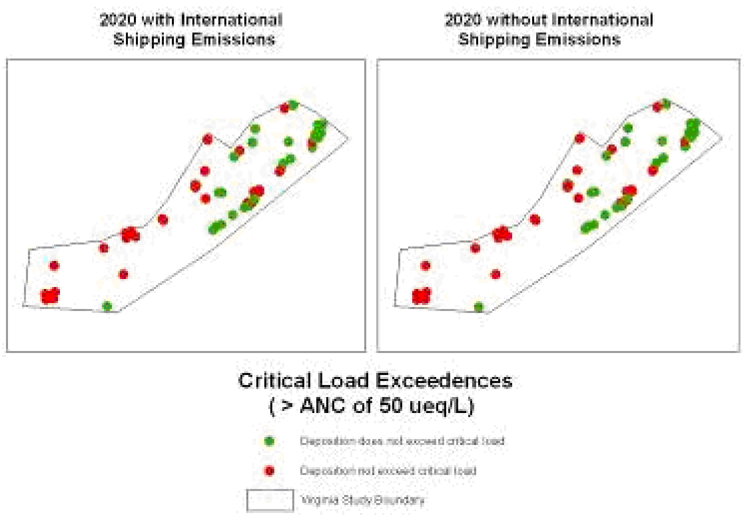


Figure 3.3-20 b . 2020; Critical Load Exceedances for ANC Concentration of 50 µeq/L. Green dots represent streams in the Virginia Blue Ridge Mountains where current nitrogen and sulfur deposition is below their critical load and maintains an ANC concentration of 50 µeq/L. Red dots are streams where current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

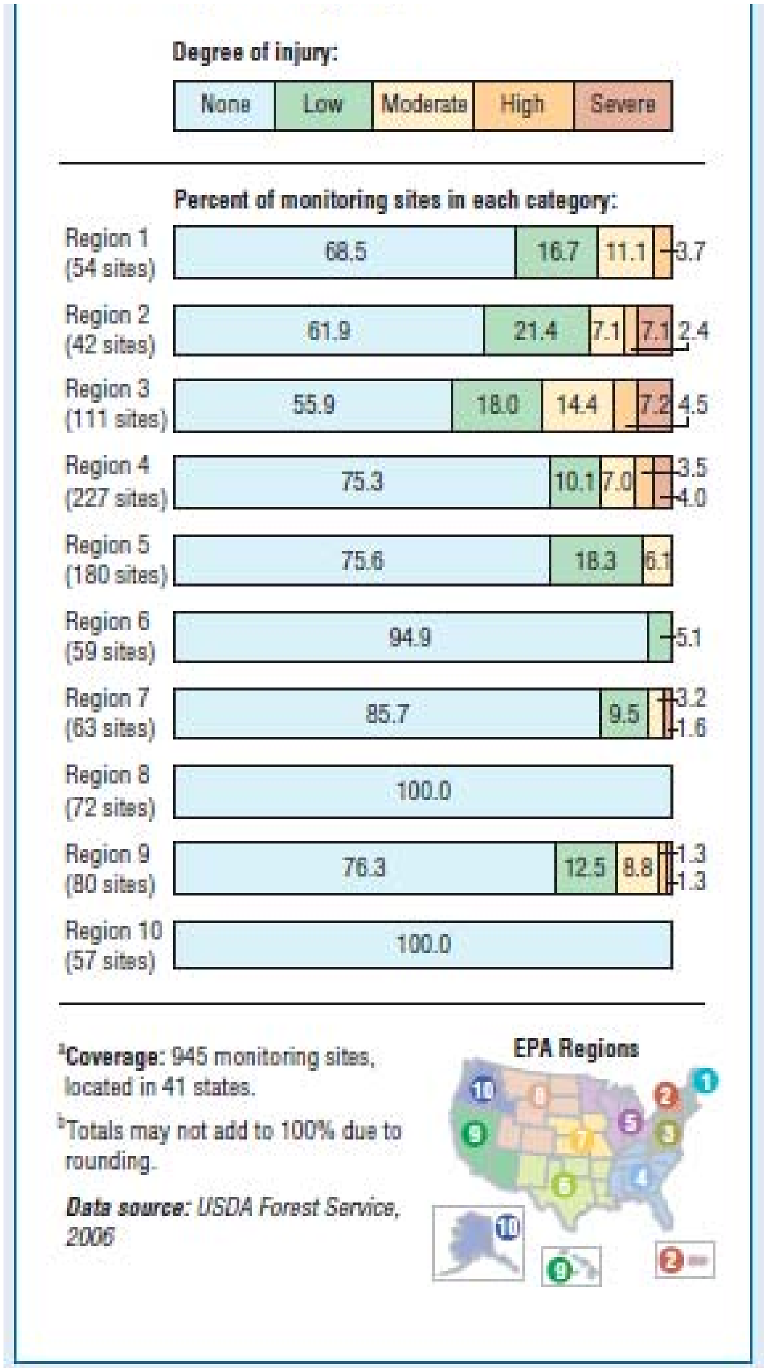


Figure 3.3-21 Ozone Injury to Forest Plants in U.S. by EPA Regions, 2002[ab]

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

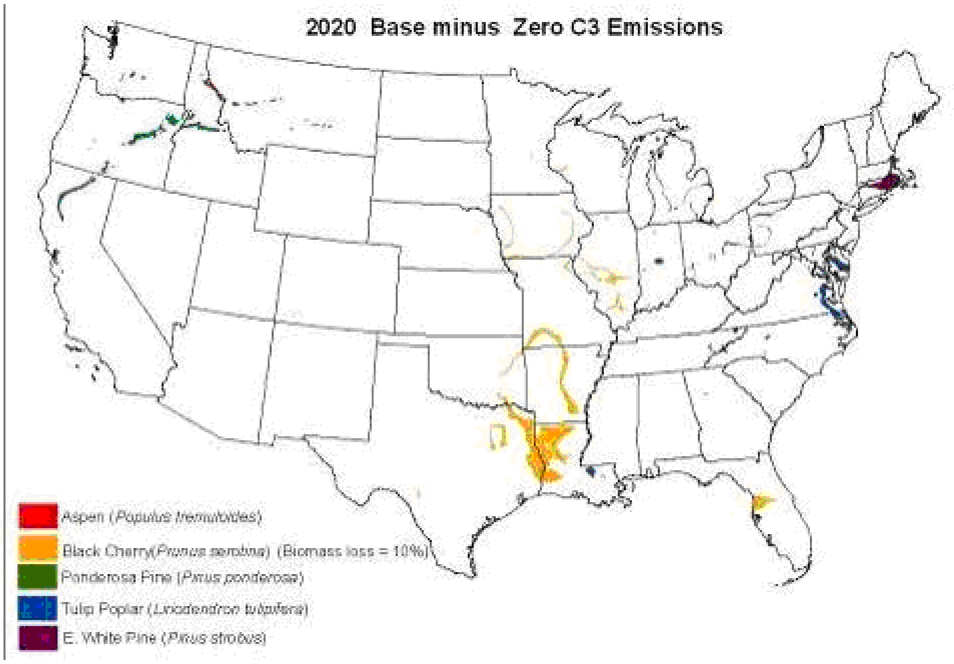


Figure 3.3-22 U.S. Geographic Areas where the Proposed ECA would Reduce Biomass Loss by More than 2%

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

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[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 μg/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

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[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).

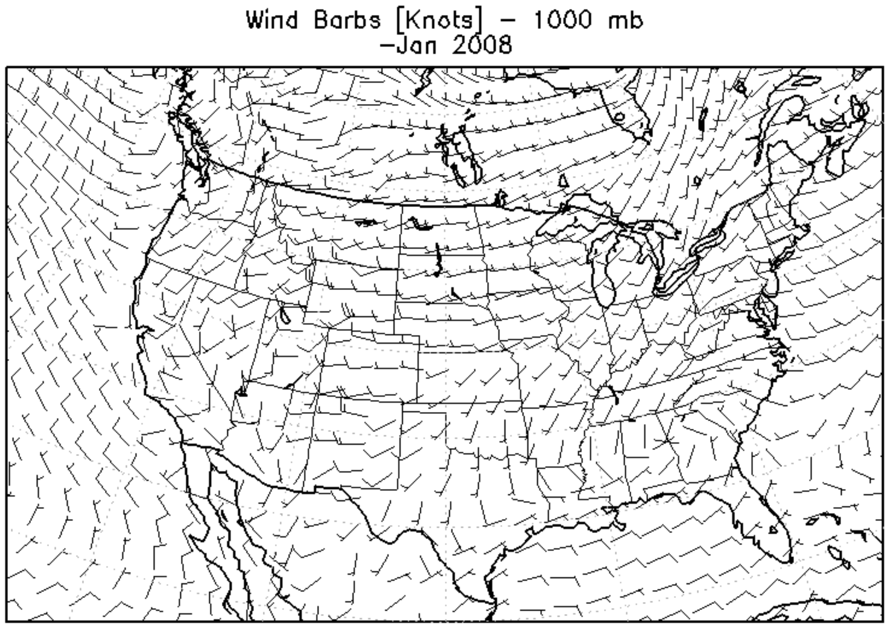

Figure 3A-1: Monthly Mean Winds in January 2008 Based on the NCEP Regional Reanalysis Dataset

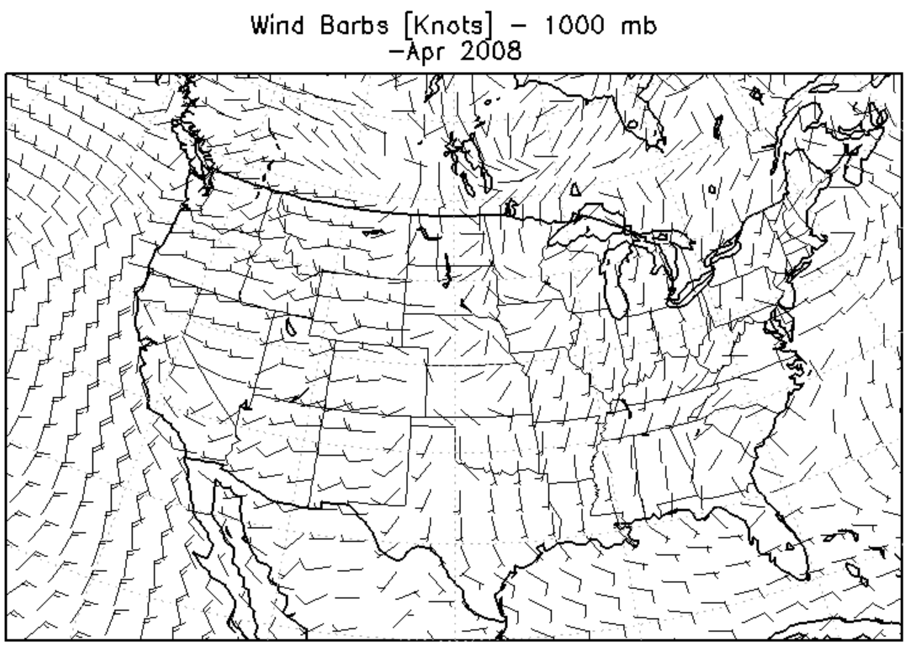

Figure 3A-2: Monthly Mean Winds in April 2008 Based on the NCEP Regional Reanalysis Dataset.

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.

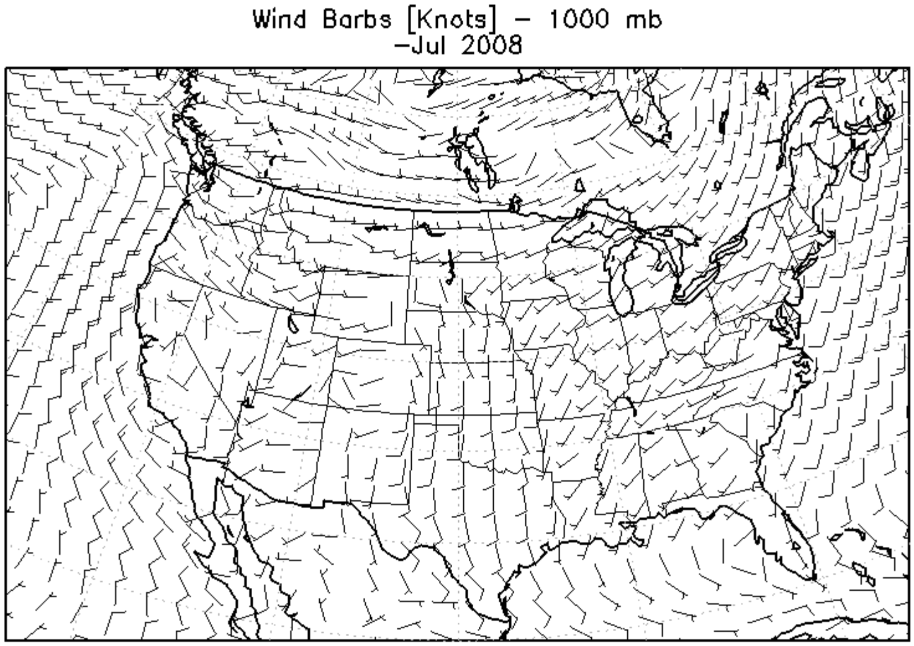

Figure 3A-3: Monthly Mean Winds in July 2008 Based on the NCEP Regional Reanalysis Dataset

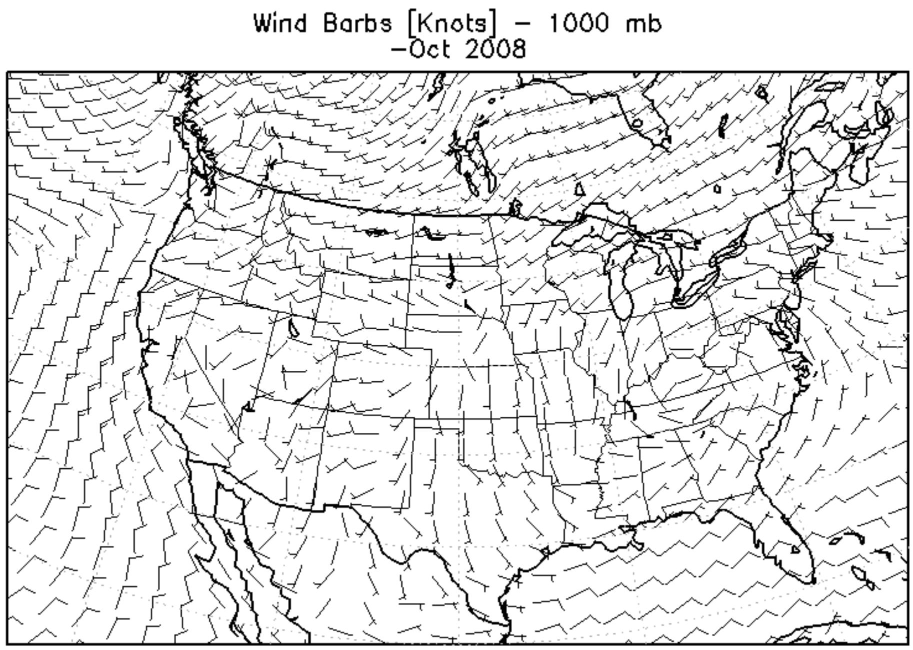

Figure 3A-4: Monthly Mean Winds in October 2008 Based on the NCEP Regional Reanalysis Dataset.

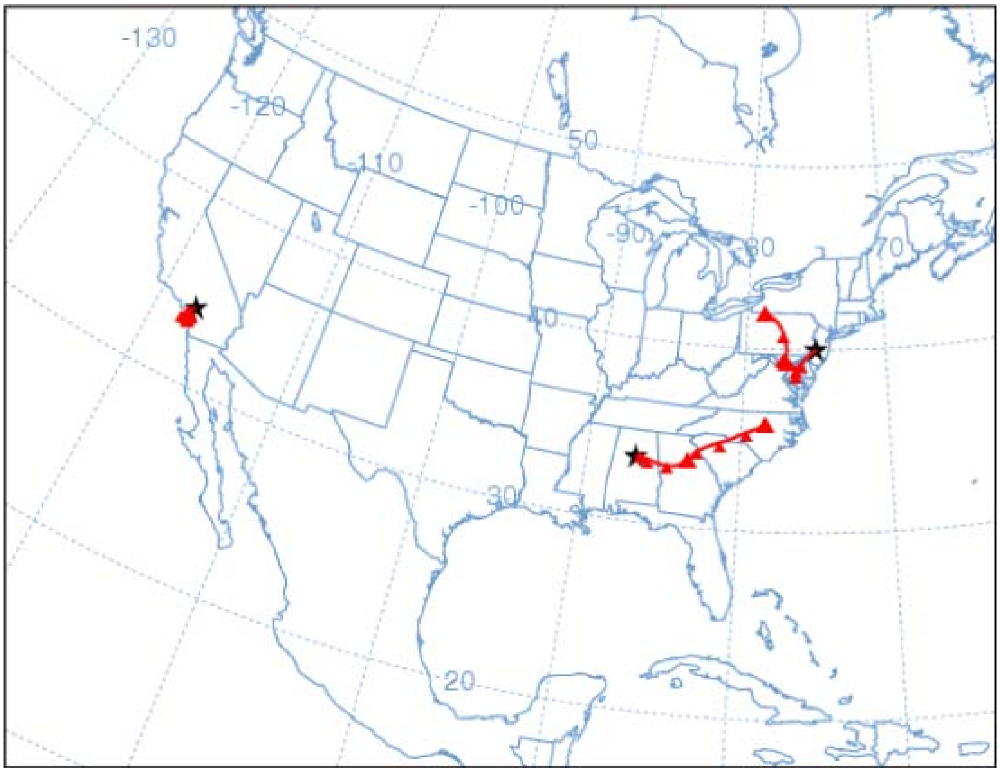

Figure 3A-5: 48-Hour Back-Trajectories from the HYSPLIT Trajectory Model. The red triangles represent how the air parcel that resided over the starred locations on 0000 GMT July 19, 2008 travelled over the preceding two days, in three hour increments.

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.

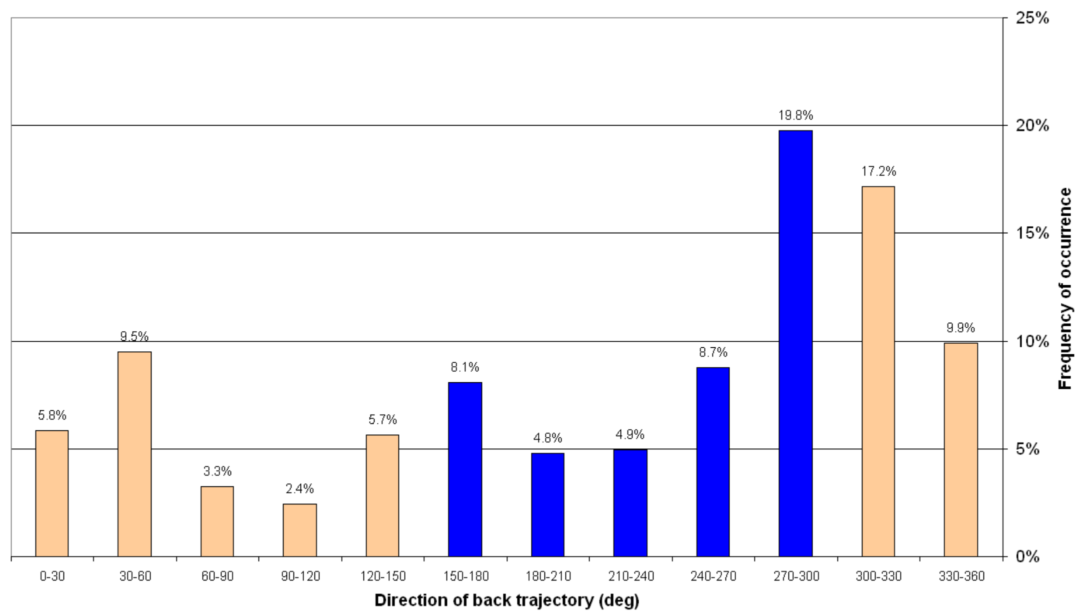

Figure 3A-6: 24-Hour Back Trajectory Directions in Los Angeles as Estimated by the HYSPLIT Model over the Period from 1995 to 2006

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.

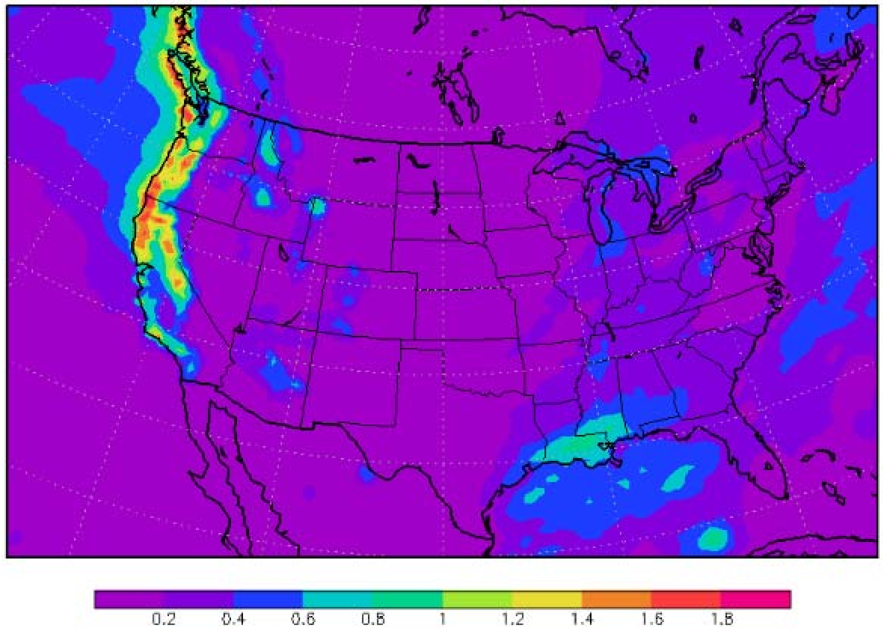

Figure 3A-8. Monthly Precipitation Accumulations in January 2008 from the NCEP Regional Reanalysis Dataset. Units are kg/m2.

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.

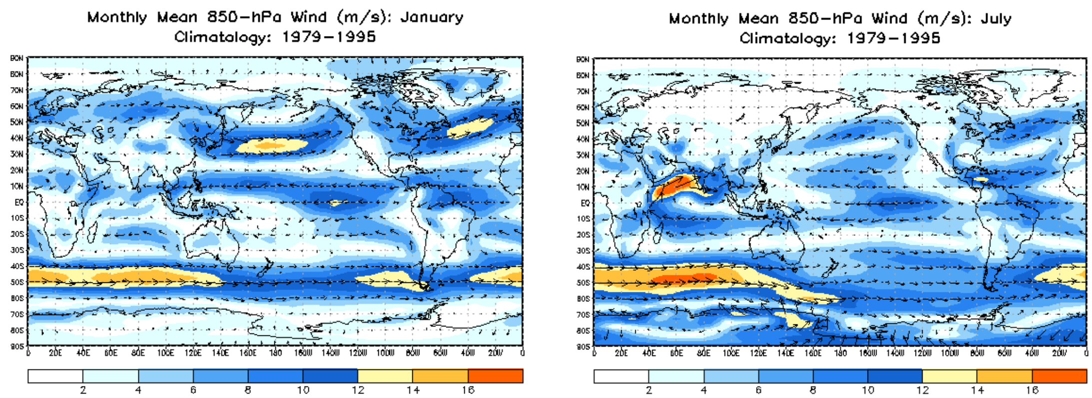

Figure 3A-9. Monthly Mean Winds at Approximately the 1,500 Meter Level in January (left) and July (right) Averaged over the Period from 1979 to 1995. Figures from NOAA Climate Prediction Center

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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4 Quantified Health Impacts Analysis

Ship emissions are responsible for a large number of adverse human health and environmental impacts, especially in densely populated coastal areas. As demonstrated in Chapters 2 and 3, ships that would operate in the proposed ECA generate emissions of NO_X (a precursor to ozone formation and secondarily-formed $PM_{2.5}$), SO_X (a precursor to secondarily-formed $PM_{2.5}$) and directly-emitted $PM_{2.5}$. These pollutants contribute to ambient concentrations of $PM_{2.5}$ and ozone that cause harm to human health and the environment. This chapter presents the U.S.-related health impacts associated with emissions from ships, both in terms of the expected contribution of overall ship emissions to adverse health impacts on land and the reductions in adverse health impacts that can be expected to occur from the adoption of the proposed ECA. Reductions in ambient $PM_{2.5}$ and ozone that will result from the proposed ECA are expected to benefit human health in the form of avoided premature deaths and other serious human health effects, as well as other important public health and environmental effects.

The most conservative premature mortality estimates (Pope et al., 2002 for $PM_{2.5}$ and Bell et al., 2004 for ozone)[1,2] suggest that implementation of the proposed ECA would reduce approximately 3,500 premature mortalities in 2020. The upper end of the premature mortality estimates (Laden et al., 2006 for $PM_{2.5}$ and Levy et al., 2005 for ozone)[3,4] suggest that implementation of the proposed ECA would increase the estimate of avoided premature mortalities to approximately 8,100 in 2020. Thus, even taking the most conservative premature mortality assumptions, the health impacts of the proposed ECA are clearly substantial.

The health impacts modeling presented in this Chapter is based on peer-reviewed studies of air quality and health and welfare effects associated with improvements in air quality. The health impact estimates for the proposed ECA are based on an analytical structure and sequence consistent with health impacts analyses performed by the United States Environmental Protection Agency (US EPA) for its recent analyses in support of the final Ozone National Ambient Air Quality Standard (NAAQS) and the final PM NAAQS as well as all of its recent mobile source emission control programs.[5,6] For a more detailed discussion of the principles of health impacts analysis used here, we refer the reader to those NAAQS documents.

Benefits estimated for this analysis were generated using the Environmental Benefits Mapping and Analysis Program (BenMAP). BenMAP is a computer program developed by the US EPA that integrates a number of modeling elements (e.g., interpolation functions, population projections, health impact functions, valuation functions, analysis and pooling methods) to translate modeled air concentration estimates into health effect incidence estimates. Interested parties may wish to consult the webpage http://www.epa.gov/ttn/ecas/benmodels.html for more information.

The general health impacts analysis framework is as follows:

- Using baseline and control emissions inventories for the emission species expected to affect ambient air quality (NO_X, SO_2, and $PM_{2.5}$; see Chapter 2), we carried out sophisticated photochemical air quality models to estimate baseline and control ambient concentrations of PM and ozone for 2020 (see Chapter 3).

- The estimated changes in ambient concentrations are then combined with monitoring data to estimate population-level potential exposures to changes in ambient concentrations for use in estimating health effects (see Chapter 3). Modeled changes in ambient data are also used to estimate changes in visibility.

- Changes in population exposure to ambient air pollution are used along with impact functions[A] to generate estimated reductions in the incidence of health effects. Because these estimates contain uncertainty, we characterize the health impact estimates probabilistically when appropriate information is available.

Table 4-1 presents the human health impacts we are able to quantify using this methodology. However, the full complement of human health and welfare effects associated with PM and ozone remains unquantified because of current limitations in methods or available data. We have not quantified a number of known or suspected health effects linked with ozone and PM for which appropriate health impact functions are not available or which do not provide easily interpretable outcomes (i.e., changes in heart rate variability). Additionally, we are unable to quantify a number of known environmental (welfare) effects, including reduced acid and particulate deposition damage to cultural monuments and other materials, and environmental benefits due to reductions of impacts of eutrophication in coastal areas. These unquantified welfare effects are also listed in Table 4-1. Both the unquantified and quantified environmental benefits of the proposed ECA are described further in Chapter 5. In sum, the health benefits quantified in this Chapter are likely underestimates of the total benefits attributable to the implementation of the proposed ECA.

[A] The term "impact function" as used here refers to the combination of a) an effect estimate obtained from the epidemiological literature, b) the baseline incidence estimate for the health effect of interest in the modeled population, c) the size of that modeled population, and d) the change in the ambient air pollution metric of interest. These elements are combined in the impact function to generate estimates of changes in incidence of the health effect. The impact function is distinct from the concentration-response (C-R) function, which strictly refers to the estimated equation from the epidemiological study relating incidence of the health effect and ambient pollution. We refer to the specific value of the relative risk or estimated coefficients in the epidemiological study as the "effect estimate." In referencing the functions used to generate changes in incidence of health effects for this analysis, we use the term "impact function" rather than C-R function because "impact function" includes all key input parameters used in the incidence calculation.

Table 4-1 Human Health and Welfare Effects of Pollutants Affected by the Proposed ECA

POLLUTANT/ EFFECT	*QUANTIFIED ESTIMATES*[A]	*UNQUANTIFIED EFFECTS - CHANGES IN:*
PM/Health[b]	Premature mortality based on both cohort study estimates [c,d] Bronchitis: chronic and acute Hospital admissions: respiratory and cardiovascular Emergency room visits for asthma Nonfatal heart attacks (myocardial infarction) Lower and upper respiratory illness Minor restricted-activity days Work loss days Asthma exacerbations (asthmatic population) Respiratory symptoms (asthmatic population) Infant mortality	Subchronic bronchitis cases Low birth weight Pulmonary function Chronic respiratory diseases other than chronic bronchitis Nonasthma respiratory emergency room visits
PM/Welfare		Value of recreational and residential visibility Household soiling
Ozone/Health[e]	Premature mortality: short-term exposures Hospital admissions: respiratory Emergency room visits for asthma Minor restricted-activity days School loss days Asthma attacks Acute respiratory symptoms	Cardiovascular emergency room visits Chronic respiratory damage[f] Premature aging of the lungs[f] Nonasthma respiratory emergency room visits
Ozone/Welfare	Decreased outdoor worker productivity Forest biomass	Yields for commercial crops Yields for commercial forests and noncommercial crops Damage to urban ornamental plants Recreational demand from damaged forest aesthetics Ecosystem functions
Nitrogen Deposition/ Welfare		Commercial forests due to acidic sulfate and nitrate deposition Commercial freshwater fishing due to acidic deposition Recreation in terrestrial ecosystems due to acidic deposition Commercial fishing, agriculture, and forests due to nitrogen deposition Recreation in estuarine ecosystems due to nitrogen deposition Ecosystem functions Passive fertilization
NO_X/Health		Lung irritation Lowered resistance to respiratory infection Hospital admissions for respiratory and cardiac diseases

[a] Primary quantified effects are those included in this analysis.

[b] In addition to primary endpoints, there are a number of biological responses that have been associated with PM and ozone health effects including morphological changes and altered host defense mechanisms. The public health impact of these biological responses may be partly represented by our quantified endpoints.

[c] Cohort estimates are designed to examine the effects of long term exposures to ambient pollution, but relative risk estimates may also incorporate some effects due to shorter term exposures (see Kunzli, 2001 for a discussion of this issue).
[d] While some of the effects of short-term exposure are likely to be captured by the cohort estimates, there may be additional premature mortality from short-term PM exposure not captured in the cohort estimates included in the primary analysis.
[e] The public health impact of biological responses such as increased airway responsiveness to stimuli, inflammation in the lung, acute inflammation and respiratory cell damage, and increased susceptibility to respiratory infection are likely partially represented by our quantified endpoints.
[f] The public health impact of effects such as chronic respiratory damage and premature aging of the lungs may be partially represented by quantified endpoints such as hospital admissions or premature mortality, but a number of other related health impacts, such as doctor visits and decreased athletic performance, remain unquantified.

4.1 Health Impacts Analysis Results for the Proposed ECA

Tables 4.1-1 and 4.1-2 present the annual $PM_{2.5}$ and ozone health impacts for two scenarios. The first scenario assesses the annual health impact of ship emissions if current levels of per-unit emissions are assumed to occur in 2020. The second scenario assesses the annual reduction of ship-related health impacts if the ECA standards are in place in 2020.

Table 4.1-1. Estimated $PM_{2.5}$-Related Health Impacts Associated with Ship Emissions[a]

Health Effect	2020 Annual Ship-Related Incidence (5^{th}% - 95^{th}%ile)	2020 Annual Reduction in Ship-Related Incidence w/ 200nm ECA (5^{th}% - 95^{th}%ile)
Premature Mortality[b]		
Adult, age 30+, ACS Cohort Study (Pope et al., 2002)	4,300 (1,700-7,000)	3,400 (1,300 – 5,500)
Adult, age 25+, Six-Cities Study (Laden et al., 2006)	9,800 (5,400-14,000)	7,800 (4,300 – 11,000)
Infant, age <1 year (Woodruff et al., 1997)	16 (0-42)	12 (0 – 33)
Chronic bronchitis (adult, age 26 and over)	4,300 (810-7,800)	3,300 (620 – 6,000)
Non-fatal myocardial infarction (adult, age 18 and over)	8,900 (4,900-13,000)	7,200 (3,900 – 10,000)
Hospital admissions - respiratory (all ages)[c]	990 (490-1,500)	780 (380 – 1,200)
Hospital admissions - cardiovascular (adults, age >18)[d]	2,100 (1,500-2,400)	1,600 (1,200 – 1,900)
Emergency room visits for asthma (age 18 years and younger)	2,500 (1,500-3,500)	1,900 (1,100 – 2,700)
Acute bronchitis, (children, age 8-12)	11,000 (0-22,000)	8,500 (0 – 17,000)
Lower respiratory symptoms (children, age 7-14)	84,000 (40,000-130,000)	66,000 (32,000 – 99,000)
Upper respiratory symptoms (asthmatic children, age 9-18)	62,000 (19,000-100,000)	48,000 (15,000 – 82,000)
Asthma exacerbation (asthmatic children, age 6-18)	79,000 (8,600-220,000)	62,000 (6,700 – 180,000)

Work loss days	580,000 (510,000-650,00)	460,000 (400,000 – 520,000)
Minor restricted activity days (adults age 18-65)	3,400,000 (2,900,000-4,000,000)	2,700,000 (2,300,000 – 3,100,000)

Notes:
[a] Incidence is rounded to two significant digits. Estimates represent incidence within the 48 contiguous United States.
[b] PM-related adult mortality based upon the American Cancer Society (ACS) Cohort Study (Pope et al., 2002) and the Six-Cities Study (Laden et al., 2006). Note that these are two alternative estimates of adult mortality and should not be summed. PM-related infant mortality based upon a study by Woodruff, Grillo, and Schoendorf, (1997).
[c] Respiratory hospital admissions for PM include admissions for chronic obstructive pulmonary disease (COPD), pneumonia and asthma.
[d] Cardiovascular hospital admissions for PM include total cardiovascular and subcategories for ischemic heart disease, dysrhythmias, and heart failure.

Table 4.1-2. Estimated Ozone-Related Health Impacts Associated with Ship Emissions[a]

Health Effect	2020 Annual Ship-Related Incidence (5th% - 95th%ile)	2020 Annual Reduction in Ship-Related Incidence w/ 200nm ECA (5th% - 95th%ile)
Premature Mortality, All ages[b]		
Multi-City Analyses		
Bell et al (2004) – Non-accidental	370 (160-570)	61 (23 – 98)
Huang et al (2005) – Cardiopulmonary	620 (290-940)	100 (43 – 160)
Schwartz, (2005) – Non-accidental	560 (240-890)	93 (34 – 150)
Meta-analyses:		
Bell et al (2005) – All cause	1,200 (660-1,700)	200 (100 – 290)
Ito et al (2005) – Non-accidental	1,600 (1,100-2,200)	270 (170 – 370)
Levy et al (2005) – All cause	1,700 (1,200-2,100)	280 (200 – 360)
Hospital admissions- respiratory causes (adult, 65 and older)[c]	2,900 (400-4,800)	470 (46 – 830)
Hospital admissions -respiratory causes (children, under 2)	2,400 (1,200-3,500)	380 (180 – 590)
Emergency room visit for asthma (all ages)	1,300 (0-3,500)	210 (0 – 550)
Minor restricted activity days (adults, age 18-65)	2,300,000 (1,100,000-3,400,000)	360,000 (160,000 – 570,000)
School absence days	810,000 (360,000-1,100,000)	130,000 (51,000 – 190,000)

[a] Incidence is rounded to two significant digits. Estimates represent incidence within the 48 contiguous United States.
[b] Estimates of ozone-related premature mortality are based upon incidence estimates derived from several alternative studies: Bell et al. (2004); Huang et al. (2005); Schwartz (2005) ; Bell et al. (2005); Ito et al. (2005); Levy et al. (2005). The estimates of ozone-related premature mortality should therefore not be summed.
[c] Respiratory hospital admissions for ozone include admissions for all respiratory causes and subcategories for COPD and pneumonia.

As can be seen in Tables 4.1-1 and 4.1-2, ship emissions contribute to large numbers of adverse health impacts within the U.S. By designating an ECA, we estimate that by 2020, emission reductions will result in major reductions in health impacts, especially those associated with PM exposure. For example, we estimate that in 2020, ships emitting at their current performance would be responsible for approximately 4,300 – 9,800 cases of premature mortality in adults (range based on the health impact function used – Pope et al., 2002 and Laden et al., 2006, respectively). Improving ship emissions to ECA standards will avoid between 3,400 – 7,800 premature deaths in 2020, a reduction of approximately 79%.

We also estimate that ships are responsible for a large number of $PM_{2.5}$-related morbidity impacts. For example, we estimate that in 2020, ships emitting at their current performance would be responsible for approximately 4,300 cases of chronic bronchitis, 8,900 non-fatal heart attacks, 5,600 hospital admissions and emergency room visits, 580,000 days of work lost, and 3,400,000 days of restricted physical activity. Improving ship emissions to ECA standards will result in the avoidance of 3,300 cases of chronic bronchitis, 7,200 non-fatal heart attacks, 4,400 hospital admissions and emergency room visits, 460,000 days of work lost, and 2,700,000 days of restricted physical activity. Again, improving to ECA standards will reduce the incidence of $PM_{2.5}$-related non-fatal health impacts associated with ships by approximately 78%.

Similarly, ship emissions contribute to adverse health impacts associated with ozone exposure. For example, we estimate that in 2020, ships emitting at their current performance would be responsible for approximately 370 – 1,700 cases of premature mortality, depending on the health impact function, 6,600 hospital admissions and emergency room visits, 810,000 days of school absence, and 2,300,000 day of restricted physical activity. Improving to ECA standards will avoid between 61 – 280 premature deaths in 2020. Furthermore, it will result in the avoidance of 1,100 hospital admissions and emergency room visits, 130,000 days of school absence, and 360,000 days of restricted physical activity.

It is clear that the avoided health impacts associated with the proposed ECA are substantial. Implementation of a North American ECA would significantly improve human health, both in terms of reduced premature mortality and avoided morbidity effects.

4.2 Methodology

4.2.1 Human Health Impact Functions

Health impact functions measure the change in a health endpoint of interest, such as hospital admissions, for a given change in ambient ozone or PM concentration. Health impact functions are derived from primary epidemiology studies, meta-analyses of multiple epidemiology studies, or expert elicitations. A standard health impact function has four components: 1) an effect estimate from a particular study; 2) a baseline incidence rate for the health effect (obtained from either the epidemiology study or a source of public health statistics such as the Centers for Disease Control); 3) the size of the potentially affected population; and 4) the estimated change in the relevant ozone or PM summary measures.

A typical health impact function might look like:

$$\Delta y = y_0 \cdot \left(e^{\beta \cdot \Delta x} - 1\right),$$

where y_0 is the baseline incidence (the product of the baseline incidence rate times the potentially affected population), β is the effect estimate, and Δx is the estimated change in the summary pollutant measure. There are other functional forms, but the basic elements remain the same. The following subsections describe the sources for each of the first three elements: size of the potentially affected populations; $PM_{2.5}$ and ozone effect estimates; and baseline incidence rates. Section 4.2.2 describes the ozone and PM air quality inputs to the health impact functions.

4.2.1.1 Potentially Affected Populations

The starting point for estimating the size of potentially affected populations is the 2000 U.S. Census block level dataset.[7] Benefits Modeling and Analysis Program (BenMAP) incorporates 250 age/gender/race categories to match specific populations potentially affected by ozone and other air pollutants. The software constructs specific populations matching the populations in each epidemiological study by accessing the appropriate age-specific populations from the overall population database. BenMAP projects populations to 2020 using growth factors based on economic projections.[8]

4.2.1.2 Effect Estimate Sources

The most significant quantifiable benefits of reducing ambient concentrations of ozone and PM are attributable to reductions in human health risks. EPA's Ozone and PM Criteria Documents[9,10] and the World Health Organization's 2003 and 2004[11,12] reports outline numerous human health effects known or suspected to be linked to exposure to ambient ozone and PM. US EPA recently evaluated the ozone and PM literature for use in the benefits analysis for the final 2008 Ozone NAAQS and final 2006 PM NAAQS analyses. We use the same literature in this analysis.

It is important to note that we are unable to separately quantify all of the possible PM and ozone health effects that have been reported in the literature for three reasons: (1) the possibility of double counting (such as hospital admissions for specific respiratory diseases versus hospital admissions for all or a sub-set of respiratory diseases); (2) uncertainties in applying effect relationships that are based on clinical studies to the potentially affected population; or (3) the lack of an established concentration-response (CR) relationship. Table 4-1 lists the possible human health and welfare effects of pollutants affected by the proposed ECA. Table 4.2-1 lists the health endpoints included in this analysis.

Table 4.2-1 Ozone- and PM-Related Health Endpoints

ENDPOINT	*POLLUTANT*	*STUDY*	*STUDY POPULATION*
Premature Mortality			
Premature mortality – daily time series	O3	Bell et al (2004) (NMMAPS study)[13] – Non-accidental Huang et al (2005)[14] - Cardiopulmonary Schwartz (2005)[15] – Non-accidental <u>Meta-analyses:</u> Bell et al (2005)[16] – All cause Ito et al (2005)[17] – Non-accidental	All ages

ENDPOINT	*POLLUTANT*	*STUDY*	*STUDY POPULATION*
		Levy et al (2005)[18] – All cause	
Premature mortality —cohort study, all-cause	$PM_{2.5}$	Pope et al. (2002)[19] Laden et al. (2006)[20]	>29 years >25 years
Premature mortality — all-cause	$PM_{2.5}$	Woodruff et al. (1997)[21]	Infant (<1 year)
Chronic Illness			
Chronic bronchitis	$PM_{2.5}$	Abbey et al. (1995)[22]	>26 years
Nonfatal heart attacks	$PM_{2.5}$	Peters et al. (2001)[23]	Adults (>18 years)
Hospital Admissions			
Respiratory	O3	Pooled estimate: Schwartz (1995) - ICD 460-519 (all resp)[24] Schwartz (1994a; 1994b) - ICD 480-486 (pneumonia)[25,26] Moolgavkar et al. (1997) - ICD 480-487 (pneumonia)[27] Schwartz (1994b) - ICD 491-492, 494-496 (COPD) Moolgavkar et al. (1997) – ICD 490-496 (COPD)	>64 years
		Burnett et al. (2001)[28]	<2 years
	$PM_{2.5}$	Pooled estimate: Moolgavkar (2003)—ICD 490-496 (COPD)[29] Ito (2003)—ICD 490-496 (COPD)[30]	>64 years
	$PM_{2.5}$	Moolgavkar (2000)—ICD 490-496 (COPD)[31]	20–64 years
	$PM_{2.5}$	Ito (2003)—ICD 480-486 (pneumonia)	>64 years
	$PM_{2.5}$	Sheppard (2003)—ICD 493 (asthma)[32]	<65 years

ENDPOINT	POLLUTANT	STUDY	STUDY POPULATION
Cardiovascular	$PM_{2.5}$	Pooled estimate: Moolgavkar (2003)—ICD 390-429 (all cardiovascular) Ito (2003)—ICD 410-414, 427-428 (ischemic heart disease, dysrhythmia, heart failure)	>64 years
	$PM_{2.5}$	Moolgavkar (2000)—ICD 390-429 (all cardiovascular)	20–64 years
Asthma-related ER visits	O3	<u>Pooled estimate:</u> Jaffe et al (2003)[33] Peel et al (2005)[34] Wilson et al (2005)[35]	 5–34 years All ages All ages
Asthma-related ER visits (con't)	$PM_{2.5}$	Norris et al. (1999)[36]	0–18 years
Other Health Endpoints			
Acute bronchitis	$PM_{2.5}$	Dockery et al. (1996)[37]	8–12 years
Upper respiratory symptoms	$PM_{2.5}$	Pope et al. (1991)[38]	Asthmatics, 9–11 years
Lower respiratory symptoms	$PM_{2.5}$	Schwartz and Neas (2000)[39]	7–14 years
Asthma exacerbations	$PM_{2.5}$	Pooled estimate: Ostro et al. (2001)[40] (cough, wheeze and shortness of breath) Vedal et al. (1998)[41] (cough)	6–18 years[a]
Work loss days	$PM_{2.5}$	Ostro (1987)[42]	18–65 years
School absence days	O3	<u>Pooled estimate:</u> Gilliland et al. (2001)[43] Chen et al. (2000)[44]	5–17 years[b]
Minor Restricted Activity Days (MRADs)	O3	Ostro and Rothschild (1989)[45]	18–65 years
	$PM_{2.5}$	Ostro and Rothschild (1989)	18–65 years

[a] The original study populations were 8 to 13 for the Ostro et al. (2001) study and 6 to 13 for the Vedal et al. (1998) study. Based on advice from the Science Advisory Board Health Effects Subcommittee (SAB-HES), we extended the applied population to 6 to 18, reflecting the common biological basis for the effect in children in the broader age group. See: U.S. Science Advisory Board. 2004. Advisory Plans for Health Effects Analysis in the Analytical Plan for EPA's Second Prospective Analysis –Benefits and Costs of the Clean Air Act, 1990—2020. EPA-SAB-COUNCIL-ADV-04-004. See also National Research Council (NRC). 2002. *Estimating the Public Health Benefits of Proposed Air Pollution Regulations*. Washington, DC: The National Academies Press.

[b] Gilliland et al. (2001) studied children aged 9 and 10. Chen et al. (2000) studied children 6 to 11. Based on recent advice from the National Research Council and the EPA SAB-HES, we have calculated reductions in school absences for all school-aged children based on the biological similarity between children aged 5 to 17.

In selecting epidemiological studies as sources of effect estimates, we applied several criteria to develop a set of studies that is likely to provide the best estimates of impacts in the U.S. To account for the potential impacts of different health care systems or underlying health status of populations, we give preference to U.S. studies over non-U.S. studies. In addition, due to the potential for confounding by co-pollutants, we give preference to effect estimates from models including both ozone and PM over effect estimates from single-pollutant models.[46,47]

4.2.1.2.1 *$PM_{2.5}$-Related Health Impact Functions*

$PM_{2.5}$-Related Adult Premature Mortality

Both long- and short-term exposures to ambient levels of air pollution have been associated with increased risk of premature mortality. The size of the mortality risk estimates from epidemiological studies, the serious nature of the effect itself, and the high monetary value ascribed to prolonging life make mortality risk reduction the most significant health endpoint quantified in this analysis.

Although a number of uncertainties remain to be addressed by continued research (NRC, 1998),[48] a substantial body of published scientific literature documents the correlation between elevated PM concentrations and increased mortality rates (US EPA, 2004).[49] Time-series methods have been used to relate short-term (often day-to-day) changes in PM concentrations and changes in daily mortality rates up to several days after a period of elevated PM concentrations. Cohort methods have been used to examine the potential relationship between community-level PM exposures over multiple years (i.e., long-term exposures) and community-level annual mortality rates. Researchers have found statistically significant associations between PM and premature mortality using both types of studies. In general, the risk estimates based on the cohort studies are larger than those derived from time-series studies. Cohort analyses are thought to better capture the full public health impact of exposure to air pollution over time, because they capture the effects of long-term exposures and possibly some component of short-term exposures (Kunzli et al., 2001; NRC, 2002).[50,51] This section discusses some of the issues surrounding the estimation of premature mortality.

Over a dozen studies have found significant associations between various measures of long-term exposure to PM and elevated rates of annual mortality, beginning with Lave and Seskin (1977).[52] Most of the published studies found positive (but not always statistically significant) associations with available PM indices such as total suspended particles (TSP). However, exploration of alternative model specifications sometimes raised questions about causal relationships (e.g., Lipfert, Morris, and Wyzga [1989]).[53] These early "ecological cross-sectional" studies (e.g., Lave and Seskin [1977]; Ozkaynak and Thurston [1987][54]) were criticized for a number of methodological limitations, particularly for inadequate control at the individual level for variables that are potentially important in causing mortality, such as wealth, smoking, and diet. Over the last 10 years, several studies using "prospective cohort" designs have been published that appear to be consistent with the earlier body of literature. These new "prospective cohort" studies reflect a significant improvement over the earlier work because they include individual-level information with respect to health status and residence. The most extensive analyses have been based on data from two prospective cohort groups, often referred to as the Harvard "Six-Cities Study" (Dockery et al., 1993;[55] Laden et al, 2006) and the "American Cancer Society or ACS study" (Pope et al., 1995;[56] Pope et al, 2002; Pope et al, 2004[57]); these studies have found consistent relationships between fine particle indicators and premature mortality across multiple locations in the United States. A third major data set comes from the California-based 7th Day Adventist Study (e.g., Abbey et al., 1999),[58] which reported associations between long-term PM exposure and mortality in men. Results from this cohort, however, have been inconsistent, and the air quality results are not geographically representative of most of the United States, and the lifestyle of the population is not reflective of much of the U.S. population. Analysis is also available for a cohort of adult male veterans diagnosed with hypertension has been examined (Lipfert et al., 2000; Lipfert et al, 2003, 2006).[59,60,61] The characteristics of this group differ from the cohorts in the Six-Cities, ACS, and 7th Day Adventist studies with respect to income, race, health status, and smoking status. Unlike previous long-term analyses, this study found some associations between mortality and ozone but found inconsistent results for PM indicators. Because of the selective nature of the population in the veteran's cohort, we have chosen not to include any effect estimates from the Lipfert et al. (2000) study in our benefits assessment.[B]

Given their consistent results and broad geographic coverage, and importance in informing the NAAQS development process, the Six-Cities and ACS data have been particularly important in benefits analyses. The credibility of these two studies is further enhanced by the fact that the initial published studies (Pope et al, 1995 and Dockery et al 1993) were subject to extensive reexamination and reanalysis by an independent team of scientific experts commissioned by the Health Effects Institute (HEI) (Krewski et al., 2000).[62] The final results of the reanalysis were then independently peer reviewed by a Special Panel of the HEI Health Review Committee. The results of these reanalyses confirmed and expanded those of the original investigators. While the HEI reexamination lends credibility to the original studies, it also highlights sensitivities concerning the relative impact of various pollutants, such as SO_2, the potential role of education in mediating the association between pollution and mortality, and the influence of spatial correlation modeling.

Further confirmation and extension of the findings of the 1993 Six City Study and the 1995 ACS study were recently completed using more recent air quality and a longer follow-up period for the ACS cohort was recently published (Pope et al, 2002, 2004; Laden et al, 2006). The follow up to the Harvard Six City Study both confirmed the effect size from the first analysis and provided additional confirmation that reductions in $PM_{2.5}$ are likely to result in reductions in the risk of premature death. This additional evidence stems from the observed reductions in $PM_{2.5}$ in each city during the extended follow-up period. Laden et al. (2006) found that mortality rates consistently went down at a rate proportionate to the observed reductions in $PM_{2.5}$.

[B] US EPA recognizes that the ACS cohort also is not representative of the demographic mix in the general population. The ACS cohort is almost entirely white and has higher income and education levels relative to the general population. US EPA's approach to this problem is to match populations based on the potential for demographic characteristics to modify the effect of air pollution on mortality risk. Thus, for the various ACS-based models, we are careful to apply the effect estimate only to ages matching those in the original studies, because age has a potentially large modifying impact on the effect estimate, especially when younger individuals are excluded from the study population. For the Lipfert analysis, the applied population should be limited to that matching the sample used in the analysis. This sample was all male, veterans, and diagnosed hypertensive. There are also a number of differences between the composition of the sample and the general population, including a higher percentage of African Americans (35%) and a much higher percentage of smokers (81% former smokers, 57% current smokers) than the general population (12% African American, 24% current smokers).

The extended analyses of the ACS cohort data (Pope et al., 2002, 2004) provides additional refinements to the analysis of PM-related mortality by a) extending the follow-up period for the ACS study subjects to 16 years, which triples the size of the mortality data set; b) substantially increasing exposure data, including additional measurement of cohort exposure to $PM_{2.5}$ following implementation of the $PM_{2.5}$ standard in 1999; c) controlling for a variety of personal risk factors including occupational exposure and diet; and d) using advanced statistical methods to evaluate specific issues that can adversely affect risk estimates including the possibility of spatial autocorrelation of survival times in communities located near each other.

For this analysis, we use the ACS study because it includes a large sample size and longer exposure interval and covers more locations (e.g., 50 cities compared to the Six-Cities Study) than other studies of its kind. The relative risks derived from the ACS study are based on the average exposure to $PM_{2.5}$, measured by the average of two $PM_{2.5}$ measurements, over the periods 1979–1983 and 1999–2000. In addition to relative risks for all-cause mortality, the ACS study provides relative risks for cardiopulmonary, lung cancer, and all-other cause mortality. Because of concerns regarding the statistical reliability of the “all-other” cause mortality relative risk estimates, we calculate mortality impacts for this analysis using the all-cause relative risk.

We also include a separate estimate based on the Six-cities study to complement the estimate based on the ACS study. We use this specific estimate because it reflects the most up-to-date science and reflects the weight that experts have placed on both the ACS and Harvard Six-city studies (see the results of the PM mortality expert elicitation).[63]

Because of the differences in the study designs and populations considered in the ACS and Harvard Six-cities studies, we do not pool the results of the studies and instead present a range of estimates reflecting the two sources of impact estimates.

A number of additional analyses have been conducted on the ACS cohort data (Jerrett et al., 2005;[64] Krewski et al., 2005;[65] Pope et al., 2004). These studies have continued to find a strong significant relationship between $PM_{2.5}$ and mortality outcomes. Specifically, much of the recent research has suggested a stronger relationship between cardiovascular mortality and lung cancer mortality with $PM_{2.5}$, and a less significant relationship between respiratory-related mortality and $PM_{2.5}$.

$PM_{2.5}$-Related Infant Mortality

Recently published studies have strengthened the case for an association between PM exposure and respiratory inflammation and infection leading to premature mortality in children under 5 years of age. Specifically, the release of the WHO Global Burden of Disease Study focusing on ambient air cites several recently published time-series studies relating daily PM exposure to mortality in children The study by Belanger et al. (2003)[66] also corroborates findings linking PM exposure to increased respiratory inflammation and infections in children. A study by Chay and Greenstone (2003)[67] found that reductions in TSP caused by the recession of 1981–1982 were related to reductions in infant mortality at the county level. With regard to the cohort study conducted by Woodruff et al. (1997),[68] we note several strengths of the study, including the use of a larger cohort drawn from a large number of metropolitan areas and efforts to control for a variety of individual risk factors in infants (e.g., maternal educational level, maternal ethnicity, parental marital status, and maternal smoking status). Based on these findings, the US EPA estimates infant mortality using an impact function developed from the Woodruff et al. (1997) study.

Chronic Bronchitis

Chronic bronchitis (CB) is characterized by mucus in the lungs and a persistent wet cough for at least 3 months a year for several years in a row. CB affects an estimated 5% of the U.S. population (American Lung Association, 1999).[69] A limited number of studies have estimated the impact of air pollution on new incidences of CB. Schwartz (1993)[70] and Abbey et al. (1995)[71] provide evidence that long-term PM exposure gives rise to the development of CB in the United States. Because the proposed ECA is expected to reduce primarily $PM_{2.5}$, this analysis uses only the Abbey et al. (1995) study, because it is the only study focusing on the relationship between $PM_{2.5}$ and new incidences of CB.

Nonfatal Myocardial Infarctions (heart attacks)

Nonfatal heart attacks have been linked with short-term exposures to $PM_{2.5}$ in the United States (Peters et al., 2001)[72] and other countries (Poloniecki et al., 1997).[73] We used a recent study by Peters et al. (2001) as the basis for the impact function estimating the relationship between $PM_{2.5}$ and nonfatal heart attacks. A more recent study by Zanobetti and Schwartz (2005)[74] used a similar method to Peters et al. (2001), but focused on adults 65 and older, and used PM_{10} as the PM indicator. They found a significant relationship between nonfatal heart attacks and PM_{10}, although the magnitude of the effect was much lower than Peters et al. This may reflect the use of PM_{10}, the more limited age range, or the less precise diagnosis of heart attack used in defining the outcome measure. Other studies, such as Domenici et al. (2006),[75] Samet et al. (2000),[76] and Moolgavkar (2000),[77] show a consistent relationship between all cardiovascular hospital admissions, including those for nonfatal heart attacks, and PM. Given the lasting impact of a heart attack on long-term health costs and earnings, we provide a separate estimate for nonfatal heart attacks. The estimate used in the analysis of the proposed ECA is based on the single available U.S. $PM_{2.5}$ effect estimate from Peters et al. (2001). The finding of a specific impact on heart attacks is consistent with hospital admission and other studies showing relationships between fine particles and cardiovascular effects both within and outside the United States. Several epidemiologic studies (Liao et al., 1999; Gold et al., 2000; Magari et al., 2001)[78,79,80] have shown that heart rate variability (an indicator of how much the heart is able to speed up or slow down in response to momentary stresses) is negatively related to PM levels. Heart rate variability is a risk factor for heart attacks and other coronary heart diseases (Carthenon et al., 2002; Dekker et al., 2000; Liao et al., 1997; Tsuji et al., 1996).[81,82,83,84] As such, significant impacts of PM on heart rate variability are consistent with an increased risk of heart attacks.

Hospital and Emergency Room Admissions

Because of the availability of detailed hospital admission and discharge records, there is an extensive body of literature examining the relationship between hospital admissions and air pollution. Because of this, many of the hospital admission endpoints use pooled impact functions based on the results of a number of studies. In addition, some studies have examined the relationship between air pollution and emergency room (ER) visits. Since most emergency room visits do not result in an admission to the hospital (the majority of people going to the emergency room is treated and return home), we treat hospital admissions and emergency room visits separately, taking account of the fraction of emergency room visits that are admitted to the hospital.

The two main groups of hospital admissions estimated in this analysis are respiratory admissions and cardiovascular admissions. There is not much evidence linking PM with other types of hospital admissions. The only type of emergency room visits that have been consistently linked to PM in the United States are asthma-related visits.

To estimate avoided incidences of $PM_{2.5}$ related cardiovascular hospital admissions in populations aged 65 and older, we use effect estimates from studies by Moolgavkar (2003)[85] and Ito (2003).[86] However, only Moolgavkar (2000)[87] provided a separate effect estimate for populations 20 to 64.[C] Total cardiovascular hospital admissions are thus the sum of the pooled estimates from Moolgavkar (2003) and Ito (2003) for populations over 65 and the Moolgavkar (2000) based impacts for populations aged 20 to 64. Cardiovascular hospital admissions include admissions for myocardial infarctions. To avoid double-counting benefits from reductions in myocardial infarctions when applying the impact function for cardiovascular hospital admissions, we first adjusted the baseline cardiovascular hospital admissions to remove admissions for myocardial infarctions.

To estimate total avoided incidences of respiratory hospital admissions, we used impact functions for several respiratory causes, including chronic obstructive pulmonary disease (COPD), pneumonia, and asthma. As with cardiovascular admissions, additional published studies show a statistically significant relationship between PM_{10} and respiratory hospital admissions. We used only those focusing on $PM_{2.5}$. Both Moolgavkar (2000) and Ito (2003) provide effect estimates for COPD in populations over 65, allowing us to pool the impact functions for this group. Only Moolgavkar (2000) provides a separate effect estimate for populations 20 to 64. Total COPD hospital admissions are thus the sum of the pooled estimate for populations over 65 and the single study estimate for populations 20 to 64. Only Ito (2003) estimated pneumonia and only for the population 65 and older. In addition, Sheppard (2003) provided an effect estimate for asthma hospital admissions for populations under age 65. Total avoided incidence of PM-related respiratory-related hospital admissions is the sum of COPD, pneumonia, and asthma admissions.

[C] Note that the Moolgavkar (2000) study has not been updated to reflect the more stringent GAM convergence criteria. However, given that no other estimates are available for this age group, we chose to use the existing study. Updates have been provided for the 65 and older population, and showed little difference. Given the very small (<5%) difference in the effect estimates for people 65 and older with cardiovascular hospital admissions between the original and reanalyzed results, we do not expect the difference in the effect estimates for the 20 to 64 population to differ significantly. As such, the choice to use the earlier, uncorrected analysis will likely not introduce much bias.

To estimate the effects of PM air pollution reductions on asthma-related ER visits, we use the effect estimate from a study of children 18 and under by Norris et al. (1999).[88] As noted earlier, there is another study by Schwartz examining a broader age group (less than 65), but the Schwartz study focused on PM_{10} rather than $PM_{2.5}$. We selected the Norris et al. (1999) effect estimate because it better matched the pollutant of interest. Because children tend to have higher rates of hospitalization for asthma relative to adults under 65, we will likely capture the majority of the impact of $PM_{2.5}$ on asthma emergency room visits in populations under 65, although there may still be significant impacts in the adult population under 65.

Acute Health Events and Work Loss Days

As indicated in Table 4.2-1, in addition to mortality, chronic illness, and hospital admissions, a number of acute health effects not requiring hospitalization are associated with exposure to ambient levels of PM. The sources for the effect estimates used to quantify these effects are described below.

Around four percent of U.S. children between the ages of 5 and 17 experience episodes of acute bronchitis annually (American Lung Association, 2002).[89] Acute bronchitis is characterized by coughing, chest discomfort, slight fever, and extreme tiredness, lasting for a number of days. According to the MedlinePlus medical encyclopedia,[D] with the exception of cough, most acute bronchitis symptoms abate within 7 to 10 days. Incidence of episodes of acute bronchitis in children between the ages of 5 and 17 were estimated using an effect estimate developed from Dockery et al. (1996).[90]

Incidences of lower respiratory symptoms (e.g., wheezing, deep cough) in children aged 7 to 14 were estimated using an effect estimate from Schwartz and Neas (2000).[91]

Because asthmatics have greater sensitivity to stimuli (including air pollution), children with asthma can be more susceptible to a variety of upper respiratory symptoms (e.g., runny or stuffy nose; wet cough; and burning, aching, or red eyes). Research on the effects of air pollution on upper respiratory symptoms has thus focused on effects in asthmatics. Incidences of upper respiratory symptoms in asthmatic children aged 9 to 11 are estimated using an effect estimate developed from Pope et al. (1991).[92]

Health effects from air pollution can also result in missed days of work (either from personal symptoms or from caring for a sick family member). Days of work lost due to $PM_{2.5}$ were estimated using an effect estimate developed from Ostro (1987).[93]

Minor restricted activity days (MRADs) result when individuals reduce most usual daily activities and replace them with less strenuous activities or rest, yet not to the point of missing work or school. For example, a mechanic who would usually be doing physical work

[D] See http://www.nlm.nih.gov/medlineplus/ency/article/000124.htm, accessed January 2002.

most of the day will instead spend the day at a desk doing paper and phone work because of difficulty breathing or chest pain. The effect of $PM_{2.5}$ and ozone on MRAD was estimated using an effect estimate derived from Ostro and Rothschild (1989).[94]

In analyzing the proposed ECA, we focused the estimation on asthma exacerbations occurring in children and excluded adults from the calculation to avoid double counting.[E] Asthma exacerbations occurring in adults are assumed to be captured in the general population endpoints such as work loss days and MRADs. Consequently, if we had included an adult-specific asthma exacerbation estimate, we would likely double-count incidence for this endpoint. However, because the general population endpoints do not cover children (with regard to asthmatic effects), an analysis focused specifically on asthma exacerbations for children (6 to 18 years of age) could be conducted without concern for double-counting.

To characterize asthma exacerbations in children, we selected two studies (Ostro et al., 2001; Vedal et al., 1998)[95,96] that followed panels of asthmatic children. Ostro et al. (2001) followed a group of 138 African-American children in Los Angeles for 13 weeks, recording daily occurrences of respiratory symptoms associated with asthma exacerbations (e.g., shortness of breath, wheeze, and cough). This study found a statistically significant association between $PM_{2.5}$, measured as a 12-hour average, and the daily prevalence of shortness of breath and wheeze endpoints. Although the association was not statistically significant for cough, the results were still positive and close to significance; consequently, we decided to include this endpoint, along with shortness of breath and wheeze, in generating incidence estimates (see below). Vedal et al. (1998) followed a group of elementary school children, including 74 asthmatics, located on the west coast of Vancouver Island for 18 months including measurements of daily peak expiratory flow (PEF) and the tracking of respiratory symptoms (e.g., cough, phlegm, wheeze, chest tightness) through the use of daily diaries. Association between PM_{10} and respiratory symptoms for the asthmatic population was only reported for two endpoints: cough and PEF. Because it is difficult to translate PEF measures into clearly defined health endpoints that can be monetized, we only included the cough-related effect estimate from this study in quantifying asthma exacerbations. We employed the following pooling approach in combining estimates generated using effect estimates from the two studies to produce a single asthma exacerbation incidence estimate. First, we pooled the separate incidence estimates for shortness of breath, wheeze, and cough generated using effect estimates from the Ostro et al. study, because each of these endpoints is aimed at capturing the same overall endpoint (asthma exacerbations) and there could be overlap in their predictions. The pooled estimate from the Ostro et al. study is then pooled

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[E] Estimating asthma exacerbations associated with air pollution exposures is difficult, due to concerns about double-counting of benefits. Concerns over double-counting stem from the fact that studies of the general population also include asthmatics, so estimates based solely on the asthmatic population cannot be directly added to the general population numbers without double-counting. In one specific case (upper respiratory symptoms in children), the only study available is limited to asthmatic children, so this endpoint can be readily included in the calculation of total benefits. However, other endpoints, such as lower respiratory symptoms and MRADs, are estimated for the total population that includes asthmatics. Therefore, to simply add predictions of asthma-related symptoms generated for the population of asthmatics to these total population-based estimates could result in double-counting, especially if they evaluate similar endpoints.

with the cough-related estimate generated using the Vedal study. The rationale for this second pooling step is similar to the first; both studies are attempting to quantify the same overall endpoint (asthma exacerbations).

Additional epidemiological studies are available for characterizing asthma-related health endpoints (the full list of epidemiological studies considered for modeling asthma-related incidence is presented in Table 4.2-2). However, we do not use these additional studies in this analysis. In particular, the Yu et al. (2000)[97] estimates show a much higher baseline incidence rate than other studies, which may lead to an overstatement of the expected impacts in the overall asthmatic population. The Whittemore and Korn (1980)[98] study did not use a well-defined endpoint, instead focusing on a respondent-defined "asthma attack." Other studies looked at respiratory symptoms in asthmatics but did not focus on specific exacerbations of asthma.

Treatment of Potential Thresholds in $PM_{2.5}$-Related Health Impact Functions

Unless specifically noted, our premature mortality benefits estimates are based on an assumed cutpoint in the premature mortality concentration-response function at 10 µg/m^3, and an assumed cutpoint of 10 µg/m^3 for the concentration-response functions for morbidity associated with short term exposure to $PM_{2.5}$. The 10 µg/m^3 threshold reflects comments from the U.S. EPA's Science Advisory Board Clean Air Science Advisory Committee (CASAC) (U.S. EPA Science Advisory Board, 2005).[99]

Table 4.2-2. Studies Examining Health Impacts in the Asthmatic Population Evaluated for Use in the Health Impacts Analysis

ENDPOINT	DEFINITION	POLLUTANT	STUDY	STUDY POPULATION
Asthma Attack Indicators				
Shortness of breath	Prevalence of shortness of breath; incidence of shortness of breath	$PM_{2.5}$	Ostro et al. (2001)	African-American asthmatics, 8–13
Cough	Prevalence of cough; incidence of cough	$PM_{2.5}$	Ostro et al. (2001)	African-American asthmatics, 8–13
Wheeze	Prevalence of wheeze; incidence of wheeze	$PM_{2.5}$	Ostro et al. (2001)	African-American asthmatics, 8–13
Asthma exacerbation	>= 1 mild asthma symptom: wheeze, cough, chest tightness, shortness of breath	PM_{10}, $PM_{1.0}$	Yu et al. (2000)	Asthmatics, 5–13
Cough	Prevalence of cough	PM_{10}	Vedal et al. (1998)	Asthmatics, 6–13
Other Symptoms/Illness Endpoints				
Upper respiratory symptoms	>= 1 of the following: runny or stuffy nose; wet cough; burning, aching, or red eyes	PM_{10}	Pope et al. (1991)	Asthmatics, 9–11
Moderate or worse asthma	Probability of moderate (or worse) rating of overall asthma status	$PM_{2.5}$	Ostro et al. (1991)	Asthmatics, all ages
Acute bronchitis	>= 1 episodes of bronchitis in the past 12 months	$PM_{2.5}$	McConnell et al. (1999)	Asthmatics, 9–15
Phlegm	"Other than with colds, does this child usually seem congested in the chest or bring up phlegm?"	$PM_{2.5}$	McConnell et al. (1999)	Asthmatics, 9–15
Asthma attacks	Respondent-defined asthma attack	$PM_{2.5}$	Whittemore and Korn (1980)	Asthmatics, all ages

4.2.1.2.2 Ozone-Related Health Impact Functions

Ozone-Related Premature Mortality

While particulate matter is the criteria pollutant most clearly associated with premature mortality, research suggests that short-term repeated ozone exposure likely contributes to premature death. In a recent report on the estimation of ozone-related premature mortality published by the National Research Council (NRC),[100] a panel of experts and reviewers concluded that ozone-related mortality should be included in estimates of the health benefits of reducing ozone exposure. The report also recommended that little or no weight be given to the assumption that there is no causal association between ozone exposure and premature mortality.

We estimate the change in mortality incidence and estimated credible interval[F] resulting from application of the effect estimate from the following studies: the Bell et al. (2004) NMMAPS analysis, Huang et al. (2004), Schwartz (2004), and effect estimates from the three meta-analyses - Bell et al. (2005), Ito et al. (2005), and Levy et al. (2005). The results from each study are presented separately to reflect differences in the study designs and assumptions about causality. However, it is important to note that this procedure only captures the uncertainty in the underlying epidemiological work, and does not capture other sources of uncertainty, such as uncertainty in the estimation of changes in air pollution exposure.

Respiratory Hospital Admissions Effect Estimates

Detailed hospital admission and discharge records provide data for an extensive body of literature examining the relationship between hospital admissions and air pollution. This is especially true for the portion of the population aged 65 and older, because of the availability of detailed Medicare records. In addition, there is one study (Burnett et al., 2001)[101] providing an effect estimate for respiratory hospital admissions in children under two.

Because the number of hospital admission studies we considered is so large, we used results from a number of studies to pool some hospital admission endpoints. Pooling is the process by which multiple study results may be combined in order to produce better estimates of the effect estimate, or β. For a complete discussion of the pooling process, see the BenMAP manual for technical details.[G] To estimate total respiratory hospital admissions associated with changes in ambient ozone concentrations for adults over 65, we first estimated the change in hospital admissions for each of the different effects categories that each study provided for each city. These cities included Minneapolis, Detroit, Tacoma and New Haven. To estimate total respiratory hospital admissions for Detroit, we added the pneumonia and COPD estimates, based on the effect estimates in the Schwartz study (1994).[102] Similarly, we summed the estimated hospital admissions based on the effect estimates the Moolgavkar study reported for Minneapolis (Moolgavkar et al., 1997).[103] To estimate total respiratory hospital admissions for Minneapolis using the Schwartz study (1994),[104] we simply estimated pneumonia hospital admissions based on the effect estimate. Making this assumption that pneumonia admissions represent the total impact of ozone on hospital admissions in this city will give some weight to the possibility that there is no relationship between ozone and COPD, reflecting the equivocal evidence represented by the different studies. We then used a fixed-effects pooling procedure to combine the two total respiratory hospital admission estimates for Minneapolis. Finally, we used random effects pooling to combine the results for Minneapolis and Detroit with results from studies in Tacoma and New Haven from Schwartz (1995).[105] As noted above, this pooling approach incorporates both the precision of the individual effect estimates and between-study variability characterizing differences across study locations.

[F] A credible interval is a posterior probability interval used in Bayesian statistics, which is similar to a confidence interval used in frequentist statistics.
[G] BenMAP and its supporting manual are available for download at http://www.epa.gov/air/benmap. Accessed January 9, 2009.

Asthma-Related Emergency Room Visits Effect Estimates

We used three studies as the source of the concentration-response functions we used to estimate the effects of ozone exposure on asthma-related emergency room (ER) visits: Peel et al. (2005);[106] Wilson et al. (2005);[107] and Jaffe et al. (2003).[108] We estimated the change in ER visits using the effect estimate(s) from each study and then pooled the results using the random effects pooling technique (see the BenMAP manual for technical details). The study by Jaffe et al. (2003) examined the relationship between ER visits and air pollution for populations aged five to 34 in the Ohio cities of Cleveland, Columbus and Cincinnati from 1991 through 1996. In single-pollutant Poisson regression models, ozone was linked to asthma visits. We use the pooled estimate across all three cities as reported in the study. The Peel et al. study (2005) estimated asthma-related ER visits for all ages in Atlanta, using air quality data from 1993 to 2000. Using Poisson generalized estimating equations, the authors found a marginal association between the maximum daily 8-hour average ozone level and ER visits for asthma over a 3-day moving average (lags of 0, 1, and 2 days) in a single pollutant model. Wilson et al. (2005) examined the relationship between ER visits for respiratory illnesses and asthma and air pollution for all people residing in Portland, Maine from 1998-2000 and Manchester, New Hampshire from 1996-2000. For all models used in the analysis, the authors restricted the ozone data incorporated into the model to the months ozone levels are usually measured, the spring-summer months (April through September). Using the generalized additive model, Wilson et al. (2005) found a significant association between the maximum daily 8-hour average ozone level and ER visits for asthma in Portland, but found no significant association for Manchester. Similar to the approach used to generate effect estimates for hospital admissions, we used random effects pooling to combine the results across the individual study estimates for ER visits for asthma. The Peel et al. (2005) and Wilson et al. (2005) Manchester estimates were not significant at the 95 percent level, and thus, the confidence interval for the pooled incidence estimate based on these studies includes negative values. This is an artifact of the statistical power of the studies, and the negative values in the tails of the estimated effect distributions do not represent improvements in health as ozone concentrations are increased. Instead these should be viewed as a measure of uncertainty due to limitations in the statistical power of the study. Note that we included both hospital admissions and ER visits as separate endpoints associated with ozone exposure, because our estimates of hospital admission costs do not include the costs of ER visits, and because most asthma ER visits do not result in a hospital admission.

Minor Restricted Activity Days Effects Estimate

Minor restricted activity days (MRADs) occur when individuals reduce most usual daily activities and replace them with less-strenuous activities or rest, but do not miss work or school. We estimated the effect of ozone exposure on MRADs using a concentration-response function derived from Ostro and Rothschild (1989).[109] These researchers estimated the impact of ozone and $PM_{2.5}$ on MRAD incidence in a national sample of the adult working population (ages 18 to 65) living in metropolitan areas. We developed separate coefficients for each year of the Ostro and Rothschild analysis (1976-1981), which we then combined for use in EPA's analysis. The effect estimate used in the impact function is a weighted average of the coefficients in Ostro and Rothschild (1989, Table 4), using the inverse of the variance as the weight.

School Absences Effect Estimate

Children may be absent from school due to respiratory or other acute diseases caused, or aggravated by, exposure to air pollution. Several studies have found a significant association between ozone levels and school absence rates. We use two studies (Gilliland et al., 2001; Chen et al., 2000)[110,111] to estimate changes in school absences resulting from changes in ozone levels. The Gilliland et al. study estimated the incidence of new periods of absence, while the Chen et al. study examined daily absence rates. We converted the Gilliland et al. estimate to days of absence by multiplying the absence periods by the average duration of an absence. We estimated 1.6 days as the average duration of a school absence, the result of dividing the average daily school absence rate from Chen et al. (2000) and Ransom and Pope (1992) by the episodic absence duration from Gilliland et al. (2001). Thus, each Gilliland et al. period of absence is converted into 1.6 absence days.

Following recent advice from the National Research Council (2002),[112] we calculated reductions in school absences for the full population of school age children, ages five to 17. This is consistent with recent peer-reviewed literature on estimating the impact of ozone exposure on school absences (Hall et al. 2003).[113] We estimated the change in school absences using both Chen et al. (2000) and Gilliland et al. (2001) and then, similar to hospital admissions and ER visits, pooled the results using the random effects pooling procedure.

4.2.1.3 Baseline PM Health Effect Incidence Rates

The epidemiological studies of the association between pollution levels and adverse health effects generally provide a direct estimate of the relationship of air quality changes to the relative risk of a health effect, rather than an estimate of the absolute number of avoided cases. For example, a typical result might be that a 10 $\mu g/m^3$ decrease in daily $PM_{2.5}$ levels might decrease hospital admissions by 3 percent. To then convert this relative change into a number of cases, the baseline incidence of the health effect is necessary. The baseline incidence rate provides an estimate of the incidence rate (number of cases of the health effect per year, usually per 10,000 or 100,000 general population) in the assessment location corresponding to baseline pollutant levels in that location. To derive the total baseline incidence per year, this rate must be multiplied by the corresponding population number (e.g., if the baseline incidence rate is number of cases per year per 100,000 population, it must be multiplied by the number of 100,000s in the population).

Some epidemiological studies examine the association between pollution levels and adverse health effects in a specific subpopulation, such as asthmatics or diabetics. In these cases, it is necessary to develop not only baseline incidence rates, but also prevalence rates for the defining condition (e.g., asthma). For both baseline incidence and prevalence data, we use age-specific rates where available. Impact functions are applied to individual age groups and then summed over the relevant age range to provide an estimate of total population benefits.

In most cases, because of a lack of data or methods, we have not attempted to project incidence rates to future years, instead assuming that the most recent data on incidence rates is the best prediction of future incidence rates. In recent years, better data on trends in incidence and prevalence rates for some endpoints, such as asthma, have become available. We are working to develop methods to use these data to project future incidence rates. However, for our primary benefits analysis, we continue to use current incidence rates. The one exception is in the case of premature mortality. In this case, we have projected mortality rates such that future mortality rates are consistent with our projections of population growth. Compared with previous analyses, this will result in a reduction in the mortality related impacts of air pollution in future years.

Table 4.2-3 summarizes the baseline incidence data and sources used in the benefits analysis. We use the most geographically disaggregated data available. For premature mortality, county-level data are available. For hospital admissions, regional rates are available. However, for all other endpoints, a single national incidence rate is used, due to a lack of more spatially disaggregated data. In these cases, we used national incidence rates whenever possible, because these data are most applicable to a national assessment of benefits. However, for some studies, the only available incidence information comes from the studies themselves; in these cases, incidence in the study population is assumed to represent typical incidence at the national level.

Table 4.2-3: Baseline Incidence Rates and Population Prevalence Rates for Use in Impact Functions, General Population

ENDPOINT	PARAMETER	RATES	
		Value	Source[a]
Mortality	Daily or annual mortality rate	Age-, cause-, and county-specific rate	CDC Wonder (1996–1998)
Hospitalizations	Daily hospitalization rate	Age-, region-, and cause-specific rate	1999 NHDS public use data files[b]
Asthma ER Visits	Daily asthma ER visit rate	Age- and region-specific visit rate	2000 NHAMCS public use data files[c]; 1999 NHDS public use data files[b]
Chronic Bronchitis	Annual prevalence rate per person - Aged 18–44 - Aged 45–64 - Aged 65 and older	 0.0367 0.0505 0.0587	1999 NHIS (American Lung Association, 2002, Table 4)
	Annual incidence rate per person	0.00378	Abbey et al. (1993, Table 3)
Nonfatal Myocardial Infarction (heart attacks)	Daily nonfatal myocardial infarction incidence rate per person, 18+ - Northeast - Midwest - South - West	 0.0000159 0.0000135 0.0000111 0.0000100	1999 NHDS public use data files[b]; adjusted by 0.93 for probability of surviving after 28 days (Rosamond et al., 1999)
Asthma Exacerbations	Incidence (and prevalence) among asthmatic African-American children - daily wheeze - daily cough - daily dyspnea	 0.076 (0.173) 0.067 (0.145) 0.037 (0.074)	Ostro et al. (2001)
	Prevalence among asthmatic children		Vedal et al. (1998)

	- daily wheeze - daily cough - daily dyspnea	0.038 0.086 0.045	
Acute Bronchitis	Annual bronchitis incidence rate, children	0.043	American Lung Association (2002, Table 11)
Lower Respiratory Symptoms	Daily lower respiratory symptom incidence among children[d]	0.0012	Schwartz et al. (1994, Table 2)
Upper Respiratory Symptoms	Daily upper respiratory symptom incidence among asthmatic children	0.3419	Pope et al. (1991, Table 2)
Work Loss Days	Daily WLD incidence rate per person (18–65) - Aged 18–24 - Aged 25–44 - Aged 45–64	 0.00540 0.00678 0.00492	1996 HIS (Adams, Hendershot, and Marano, 1999, Table 41); U.S. Bureau of the Census (2000)
Minor Restricted-Activity Days	Daily MRAD incidence rate per person	0.02137	Ostro and Rothschild (1989, p. 243)

[a] The following abbreviations are used to describe the national surveys conducted by the National Center for Health Statistics: HIS refers to the National Health Interview Survey; NHDS—National Hospital Discharge Survey; NHAMCS—National Hospital Ambulatory Medical Care Survey.

[b] See ftp://ftp.cdc.gov/pub/Health_Statistics/NCHS/Datasets/NHDS/.

[c] See ftp://ftp.cdc.gov/pub/Health_Statistics/NCHS/Datasets/NHAMCS/.

[d] Lower respiratory symptoms are defined as two or more of the following: cough, chest pain, phlegm, and wheeze.

Baseline age, cause, and county-specific mortality rates were obtained from the U.S. Centers for Disease Control and Prevention (CDC) for the years 1996 through 1998. CDC maintains an online data repository of health statistics, CDC Wonder, accessible at http://wonder.cdc.gov/. The mortality rates provided are derived from U.S. death records and U.S. Census Bureau postcensal population estimates. Mortality rates were averaged across 3 years (1996 through 1998) to provide more stable estimates. When estimating rates for age groups that differed from the CDC Wonder groupings, we assumed that rates were uniform across all ages in the reported age group. For example, to estimate mortality rates for individuals ages 30 and up, we scaled the 25- to 34-year-old death count and population by one-half and then generated a population-weighted mortality rate using data for the older age groups.

To estimate age- and county-specific mortality rates in years 2000 through 2020, we calculated adjustment factors, based on a series of Census Bureau projected national mortality rates, to adjust the CDC Wonder age- and county-specific mortality rates in 1996-1998 to corresponding rates for each future year. For the analysis year 2020, these adjustment factors ranged across age categories from 0.76 to 0.86

For the set of endpoints affecting the asthmatic population, in addition to baseline incidence rates, prevalence rates of asthma in the population are needed to define the applicable population. Table 4.2-3 lists the baseline incidence rates and their sources for asthma symptom endpoints. Table 4.2-4 lists the prevalence rates used to determine the applicable population for asthma symptom endpoints. Note that these reflect current asthma prevalence and assume no change in prevalence rates in future years.

Table 4.2-4. Asthma Prevalence Rates Used to Estimate Asthmatic Populations in Impact Functions

POPULATION GROUP	ASTHMA PREVALENCE RATES	
	Value	Source
All Ages	0.0386	American Lung Association (2002, Table 7)—based on 1999 HIS
< 18	0.0527	American Lung Association (2002, Table 7)—based on 1999 HIS
5–17	0.0567	American Lung Association (2002, Table 7)—based on 1999 HIS
18–44	0.0371	American Lung Association (2002, Table 7)—based on 1999 HIS
45–64	0.0333	American Lung Association (2002, Table 7)—based on 1999 HIS
65+	0.0221	American Lung Association (2002, Table 7)—based on 1999 HIS
Male, 27+	0.021	2000 HIS public use data files[a]
African American, 5 to 17	0.0726	American Lung Association (2002, Table 9)—based on 1999 HIS
African American, <18	0.0735	American Lung Association (2002, Table 9)—based on 1999 HIS

[a] See ftp://ftp.cdc.gov/pub/Health_Statistics/NCHS/Datasets/NHIS/2000/.

4.2.1.4 Baseline Incidence Rates for Ozone-related Health Impacts

Epidemiological studies of the association between pollution levels and adverse health effects generally provide a direct estimate of the relationship of air quality changes to the *relative risk* of a health effect, rather than estimating the absolute number of avoided cases. For example, a typical result might be that a 100 ppb decrease in daily ozone levels might, in turn, decrease hospital admissions by 3 percent. The baseline incidence of the health effect is necessary to convert this relative change into a number of cases. A baseline incidence rate is the estimate of the number of cases of the health effect per year in the assessment location, as it corresponds to baseline pollutant levels in that location. To derive the total baseline incidence per year, this rate must be multiplied by the corresponding population number. For example, if the baseline incidence rate is the number of cases per year per 100,000 people, that number must be multiplied by the number of 100,000s in the population.

Table 4.2-5 summarizes the sources of baseline incidence rates and provides average incidence rates for the endpoints included in the analysis. For both baseline incidence and prevalence data, we used age-specific rates where available. We applied concentration-response functions to individual age groups and then summed over the relevant age range to provide an estimate of total population benefits. In most cases, we used a single national incidence rate, due to a lack of more spatially disaggregated data. Whenever possible, the national rates used are national averages, because these data are most applicable to a national assessment of benefits. For some studies, however, the only available incidence information comes from the studies themselves; in these cases, incidence in the study population is assumed to represent typical incidence at the national level. Regional incidence rates are available for hospital admissions, and county-level data are available for premature mortality. We have projected mortality rates such that future mortality rates are consistent with our projections of population growth.

Table 4.2-5. National Average Baseline Incidence Rates[a]

ENDPOINT	SOURCE	NOTES	RATE PER 100 PEOPLE PER YEAR[D] BY AGE GROUP						
			<18	*18-24*	*25-34*	*35-44*	*45-54*	*55-64*	*65+*
Mortality	CDC Compressed Mortality File, accessed through CDC Wonder (1996-1998)	non-accidental	0.03	0.02	0.06	0.15	0.38	1.01	4.94
Respiratory Hospital Admissions.	1999 NHDS public use data files[b]	incidence	0.04	0.08	0.21	0.68	1.93	4.40	11.63
Asthma ER visits	2000 NHAMCS public use data files[c]; 1999 NHDS public use data files[b]	incidence	1.01	1.09	0.75	0.44	0.35	0.43	0.23
Minor Restricted Activity Days (MRADs)	Ostro and Rothschild (1989, p. 243)	incidence	–	780	780	780	780	780	–
School Loss Days	National Center for Education Statistics (1996) and 1996 HIS (Adams et al., 1999, Table 47); estimate of 180 school days per year	all-cause	990	–	–	–	–	–	–

[a] The following abbreviations are used to describe the national surveys conducted by the National Center for Health Statistics: HIS refers to the National Health Interview Survey; NHDS - National Hospital Discharge Survey; NHAMCS - National Hospital Ambulatory Medical Care Survey.
[b] See ftp://ftp.cdc.gov/pub/Health_Statistics/NCHS/Datasets/NHDS/
[c] See ftp://ftp.cdc.gov/pub/Health_Statistics/NCHS/Datasets/NHAMCS/
[d] All of the rates reported here are population-weighted incidence rates per 100 people per year. Additional details on the incidence and prevalence rates, as well as the sources for these rates are available upon request.

Table 4.2-5. National Average Baseline Incidence Rates (continued)

ENDPOINT	SOURCE	NOTES		RATE PER 100 PEOPLE PER YEAR
Asthma Exacerbations	Ostro et al. (2001)	Incidence (and prevalence) among asthmatic African-American children	Daily wheeze Daily cough Daily dyspnea	0.08 (0.17) 0.07 (0.15) 0.04 (0.07)
	Vedal et al. (1998)	Incidence (and prevalence) among asthmatic children	Daily wheeze Daily cough Daily dyspnea	0.04 0.09 0.05

4.2.2 Manipulating Air Quality Modeling Data for Health Impacts Analysis

In Chapter 3, we summarized the methods for and results of estimating air quality for the 2020 base case and proposed ECA scenario. These air quality results are in turn associated with human populations to estimate changes in health effects. For the purposes of this analysis, we focus on the health effects that have been linked to ambient changes in ozone and $PM_{2.5}$ related to emission reductions estimated to occur due to the proposed ECA. We estimate ambient $PM_{2.5}$ and ozone concentrations using the Community Multiscale Air Quality model (CMAQ). This section describes how we converted the CMAQ modeling output into full-season profiles suitable for the health impacts analysis.

4.2.2.1 General Methodology

First, we extracted hourly, surface-layer PM and ozone concentrations for each grid cell from the standard CMAQ output files. For ozone, these model predictions are used in conjunction with the observed concentrations obtained from the Aerometric Information Retrieval System (AIRS) to generate ozone concentrations for the entire ozone season.[H,I] The predicted changes in ozone concentrations from the future-year base case to future-year control scenario serve as inputs to the health and welfare impact functions of the benefits analysis (i.e., BenMAP).

To estimate ozone-related health effects for the contiguous United States, full-season ozone data are required for every BenMAP grid-cell. Given available ozone monitoring data, we generated full-season ozone profiles for each location in two steps: (1) we combined monitored observations and modeled ozone predictions to interpolate hourly ozone concentrations to a grid of 12-km by 12-km population grid cells for the contiguous 48 states, and (2) we converted these full-season hourly ozone profiles to an ozone measure of interest, such as the daily 8-hour maximum.[J,K]

For $PM_{2.5}$, we also use the model predictions in conjunction with observed monitor data. CMAQ generates predictions of hourly PM species concentrations for every grid. The species include a primary coarse fraction (corresponding to PM in the 2.5 to 10 micron size range), a primary fine fraction (corresponding to PM less than 2.5 microns in diameter), and several secondary particles (e.g., sulfates, nitrates, and organics). $PM_{2.5}$ is calculated as the sum of the primary fine fraction and all of the secondarily formed particles. Future-year estimates of $PM_{2.5}$ were calculated using relative reduction factors (RRFs) applied to 2002 ambient $PM_{2.5}$ and $PM_{2.5}$ species concentrations. A gridded field of $PM_{2.5}$ concentrations was created by interpolating Federal Reference Monitor ambient data and IMPROVE ambient data. Gridded fields of $PM_{2.5}$ species concentrations were created by interpolating US EPA speciation network (ESPN) ambient data and IMPROVE data. The ambient data were interpolated to the CMAQ 12 km grid.

The procedures for determining the RRFs are similar to those in US EPA's draft guidance for modeling the $PM_{2.5}$ standard (EPA, 1999). The guidance recommends that model predictions be used in a relative sense to estimate changes expected to occur in each major $PM_{2.5}$ species. The procedure for calculating future-year $PM_{2.5}$ design values is called

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[H] The ozone season for this analysis is defined as the 5-month period from May to September.
[I] Based on AIRS, there were 961 ozone monitors with sufficient data (i.e., 50 percent or more days reporting at least nine hourly observations per day [8 am to 8 pm] during the ozone season).
[J] The 12-km grid squares contain the population data used in the health benefits analysis model, BenMAP.
[K] This approach is a generalization of planar interpolation that is technically referred to as enhanced Voronoi Neighbor Averaging (EVNA) spatial interpolation. See the BenMAP manual for technical details, available for download at http://www.epa.gov/air/benmap.

the "Speciated Modeled Attainment Test (SMAT)." EPA used this procedure to estimate the ambient impacts of the proposed ECA controls.

4.2.2.2 Emissions Inventory Boundary Distance Error

As noted in Appendix 2F to Chapter 2, the air quality modeling used for this analysis is based on inventory estimates that were modeled using incorrect boundary information. The impact of this difference, while modest, leads to an underestimate of the benefits that are presented in this Chapter. Please refer to Appendix 2F for more information on the emissions excluded from the health impacts analysis of the proposed ECA.

4.3 Methods for Describing Uncertainty

For this analysis, consistent with the approach used in the analyses for the recent PM and Ozone NAAQS, we addressed key sources of uncertainty through Monte Carlo propagation of uncertainty in the concentration-response (CR) functions. It should be noted that the Monte Carlo-generated distributions of health impacts reflect only some of the uncertainties in the input parameters. Uncertainties associated with emissions, air quality modeling, populations, and baseline health effect incidence rates are not represented in the distributions of avoided health impacts associated with the implementation of the proposed ECA. A complete description of uncertainty related to health impacts analyses can be found in the regulatory impact analysis drafted in support of the final Ozone NAAQS analysis.[114]

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5 Costs

The reduction of SO_X, NO_X, and PM emissions from ships has an associated cost that reaches not only to the shipping industry but also to marine fuel suppliers and companies who rely on the shipping industry. Though these cost impacts do exist, analyses presented in this document indicate that the costs associated with the proposed ECA are expected to have a minimal economic impact and to be relatively small compared to the resulting improvements in air quality. This chapter describes the analyses used to evaluate the cost impacts of Tier III NO_X requirements combined with the use of lower sulfur fuel on vessels operating within the U.S. portion of the proposed ECA; including estimates of lower sulfur fuel production costs, engine and vessel hardware costs, and the associated differential operating costs. This chapter also presents cost per ton estimates for ECA-based NO_X and fuel sulfur standards and compares these costs with established land-based control programs.

The costs presented here are based on the application of ECA controls and compliance with ECA standards in 2020. Consistent with the presentation of the inventory (Chapter 2) and the benefits (Chapter 4), the estimated costs are reported for the year 2020. In this year, only new vessels will incur hardware costs, while all vessels (new or existing) will incur additional operating costs in the proposed ECA (e.g. the use of urea on an SCR equipped vessel built in or prior to 2020). A separate analysis is provided for the benefit of ship owners, which presents the estimated one-time hardware costs that may be incurred by some existing vessels to accommodate the use of lower sulfur fuel. These costs are expected to be incurred by 2015 when the fuel sulfur standards take effect, and are not included in the 2020 total. All costs are presented in terms of 2006 U.S. dollars.

5.1 Fuel Production Costs

This section presents our analysis of the impact of the proposed ECA on marine fuel costs. Distillate fuel will likely be needed to meet the 0.1 percent fuel sulfur limit, beginning in 2015, for operation in ECAs.[A] As such, the primary cost of the fuel sulfur limit will be that associated with switching from heavy fuel oil to higher-cost distillate fuel, when operating in the ECA. Some engines already operate on distillate fuel and would not be affected by fuel switching costs. Distillate fuel costs may be affected by the need to further refine the distillate fuel to meet the 0.1 percent fuel sulfur limit. To investigate these effects, studies were performed on the impact of a U.S./Canada ECA on global fuel production and costs. These studies, which are summarized below, include economic modeling to project bunker fuel demand and refinery modeling to assess the impact of a U.S./Canada ECA on fuel costs.

[A] As an alternative, an exhaust gas cleaning device (scrubber) may be used. This analysis does not include the effect on distillate fuel demand of this alternative approach. It is expected that scrubbers would only be used in the case where the operator determines that the use of a scrubber would result in a cost savings relative to using distillate fuel. Therefore we are only estimating the cost of compliance using distillate fuel here as we believe this is the most likely approach.

5.1.1 Bunker Fuel Demand Modeling

To assess the affect of an ECA on the refining industry, we needed to first understand and characterize the fuels market and more specifically the demand for the affected marine fuels both currently and in the future. Research Triangle Institute (RTI) was contracted to conduct a fuels study using an activity-based economic approach.[1] The RTI study established baseline bunker fuel demand, projected a growth rate for bunker fuel demand, and established future bunker fuel demand volumes. The basis for this work was the Global Insights economic model which projects international trade for different categories of commodities. Demand for marine fuels was derived from the demand of transportation of various types of cargoes by ship, which, in turn, was derived from the demand for commodities produced in one region of the world and consumed in another. The flow of commodities was matched with typical vessels for that trade (characterized according to size, engine power, age, specific fuel consumption, and engine load factors). Typical voyage parameters were assigned, including average ship speed, round trip mileage, tonnes of cargo shipped, and days in port. Fuel consumption for each trade route and commodity type was thus a function of commodity projections, ship characteristics, and voyage characteristics.

The bunker demand model included operation off the coasts of the contiguous United States and southeastern Alaska. The bunker demand volumes for this modeling in the Canadian portion of the ECA was based on fuel consumed by ships en route to and from Canadian ports based on estimates from Environment Canada.

These affected fuel volumes which are used in the WORLD model described below, are slightly higher than what we now estimate for the proposed ECA. This difference is because the RTI evaluation of affected fuel volumes was performed before the ECA was defined and was performed independently of the emission inventory modeling described in Chapter 2. However, we believe it is reasonable to use the fuel cost increases, on a per-tonne basis, from the WORLD modeling to estimate the impact of the proposed ECA. In earlier work,[2] EnSys modeled a number of fuel control scenarios where the volume of affected fuel was adjusted to represent 1) different ECAs or 2) various penetration scenarios of exhaust gas scrubbers (as an alternative to fuel switching). This work suggests that the differences in fuel volume between these scenarios have only a small effect on fuel cost. Although this earlier work was based on the older crude oil and refinery costs used in the expert group study, it is sufficient for observing the sensitivity of fuel cost increases to small changes (on a global scale) in affected fuel volume. In addition, the larger affected fuel volume, used in the WORLD modeling, directionally increases the projected fuel cost increases, and therefore allows for a conservative analysis.

5.1.2 Bunker Fuel Cost Modeling

5.1.2.1 Methodology

To assess the impacts of the proposed ECA on fuel costs, the World Oil Refining Logistics and Demand (WORLD) model was run by Ensys Energy & Systems, the owner and developer of the refinery model. The WORLD model is the only such model currently developed for this purpose, and was developed by a team of international petroleum consultants. It has been widely used by industries, government agencies, and OPEC over the past 13 years, including the Cross Government/Industry Scientific Group of Experts, established to evaluate the effects of the different fuel options proposed under the revision of MARPOL Annex VI.[3] The model incorporates crude sources, global regions, refinery operations, and world economics. The results of the WORLD model have been shown to be comparable to other independent predictions of global fuel, air pollutant emissions and economic predictions.

WORLD is a comprehensive, bottom-up model of the global oil downstream that includes crude and noncrude supplies; refining operations and investments; crude, products, and intermediates trading and transport; and product blending/quality and demand. Its detailed simulations are capable of estimating how the global system can be expected to operate under a wide range of different circumstances, generating model outputs such as price effects and projections of refinery operations and investments.

5.1.2.2 Assessment of the Impact of Marine Fuel Standards

During the development of the amendments to MARPOL Annex VI, a Cross Government/Industry Scientific Group of Experts was established, by IMO, to evaluate the effects of the different fuel options that were under consideration at the time. This expert group engaged the services of EnSys to assess the impact of these fuel options using the WORLD model. The final report from this study presents great detail on the capabilities of the WORLD model and provides support for why the WORLD model was chosen as the appropriate tool for modeling the economic impacts of the different fuel options.[4] The following description of the WORLD model is taken from the expert group study:

WORLD is a linear programming model that simulates the activities and economics of the world regional petroleum industry against short, medium or long term horizons. It models and captures the interactions between:

- crude supply;
- non-crudes supply: Natural gas Liquids (NGLs), merchant MTBE, biofuels, petrochemical returns, Gas To Liquid fuels (GTLs), Coal to Liquid fuels (CTLs);
- refining operations;
- refining investment;
- transportation of crudes, products and intermediates;
- product blending/quality;
- product demand; and
- market economics and pricing.

The model includes a database representing over 180 world crude oils and holds detailed, tested, state-of-the-art representation of fifty-plus refinery processes. These representations include energy requirements based on today's construction standards for new refinery units. It allows for advanced representation of processes for reformulated, ultra-lower sulfur/aromatics fuels and was extended for detailed modeling of marine fuels for the aforementioned EPA and API studies. The model contains detailed representations of the blending and key quality specifications for over 50 different products spread across the product spectrum and including multiple grades of gasolines, diesel fuels/gasoils (marine and non-marine) and residual fuels (marine and non-marine).

The refining industry is a co-product industry. This means that changes in production of one product also affect production volume and/or production costs of other products. As necessary, the model will adjust refinery throughputs and operations, crude and product trade patterns to ensure that a specified product demand slate is met, without surplus or deficit of any product.

To evaluate the impact of changes to marine fuels specifications as a result of any of the options under consideration, the model is run with a future demand scenario for all products. The first run, the base case, assumes marine fuels in line the current Annex VI regulation. The second run is done with marine fuel specifications in line with the option under consideration. Both runs are optimized independently. Since the only thing that is altered between the cases is the change in the projected marine fuels regulation, the difference between both cases is therefore a true assessment of the actual cost and other implications of the change to the marine fuels requirements under consideration. Thus, the incremental refining investment costs, incremental marine fuel costs and incremental refinery/net CO_2 emissions are all directly attributable to - and must be allocated to – the change in regulation.

Prior to the expert group study, EnSys made updates to the WORLD model to be able to perform the analysis of the impacts of different marine fuel options. As part of this effort, the refinery data, capacity additions, technology assumptions, and costs were reviewed. EnSys reviewed relevant regulations to ensure that the WORLD model was correctly positioned to undertake future analyses of marine fuels ECAs. In developing these updates, a number of issues had to be considered:

- the costs of refining, including the capital expenditures required to reduce bunker fuel sulfur content and the potential for process technology improvements;
- likely market reactions to increased bunker fuel costs, such as fuel grade availability, impacts on the overall transportation fuels balance, and competition with land-based diesel and residual fuels for feedstocks that can upgrade fuels;
- the effects of emissions trading; and
- the potential for low- and high-sulfur grade bunker sources and consumption to partially shift location depending on supply volume, potential, and economics.

The analytical system thus had to be set up to allow for alternative compliance scenarios, particularly with regard to (a) adequately differentiating bunker fuel grades; (b) allowing for differing degrees to which the ECA or other standards in a region were presumed to be met by bunker fuel sulfur reductions, rather than by other means such as scrubbing or emissions trading; and (c) allowing for all residual fuel bunker demand to be reallocated to marine diesel. Beyond any international specifications, the analytical system needed to be able to accommodate future consideration of regional, national, and local specifications.

The primary approach taken to manage these issues was to:

- expand the number of bunker grades in the model to three distillates and four residual grades;[B]
- allow for variation where necessary in (regional) sulfur standards on specific bunker grades; and
- enable residual bunker demand to be switched to marine diesel.

Other updates to the WORLD model included product transportation matrices covering tanker, interregional pipeline, and minor modes were expanded to embody the additional distillate and residual bunker grades, adjustments to the yield patterns of the residuum desulfurization, and blocking of paraffinic streams from residual fuel blends. The details of compliance in any particular region must be estimated external to the main WORLD model. As discussed above, we provided our estimates of affected fuel volumes to Ensys.

5.1.2.3 Updates for ECA Analysis

To determine the impact of the proposed ECA, the WORLD model was employed using the same basic approach as for the IMO expert group study. Modeling was performed for 2020 in which the control case included a fuel sulfur level of 0.1 percent in the U.S. and Canadian EEZs.[5] The baseline case was modeled as "business as usual" in which ships continue to use the same fuel as today. This approach was used for two primary reasons. First, significant emission benefits are expected in an ECA, beginning in 2015, due to the use of 0.1 percent sulfur fuel. These benefits, and costs, would be much higher in the early years of the program before the 0.5 percent fuel sulfur global standard goes into effect. By modeling this scenario, we are able to observe the impact of the proposed ECA in these early years. Second, there is no guarantee that the global 0.5 percent fuel sulfur standards will begin in 2020. The global standard may be delayed until 2025, subject to a fuel availability review in 2018. In addition, the 3.5 percent fuel sulfur global standard, which begins in 2012, is higher than the current residual fuel sulfur average of 2.7 percent.

In the modeling for the expert group study, crude oil prices were based on projections released by the U.S. Energy Information Administration (EIA) in 2006.[6] Since that time, oil prices have fluctuated greatly. Using new information, EIA has updated its projections of oil price for 2020.[7,8] In response to this real-world effect, the ECA modeling was conducted using the updated oil price estimates. Specifically, we used a crude oil price of $51.55 for the reference case, and $88.14/bbl for the high price case, both expressed in real (2006) dollars. These crude oil prices were input to the WORLD model which then computed residual and distillate marine oil prices for 2020. The net refinery capital impacts are imputed based on the differences in the costs to the refining industry that occur between the Base Cases and ECA cases in 2020. The

[B] Specifically, the following seven grades were implemented: MGO, plus distinct high- and low-sulfur blends for MDO and the main residual bunker grades IFO 180 and IFO 380. The latest international specifications applying to these fuels were used, as were tighter sulfur standards for the low-sulfur grades applicable in SECAs.

incremental global refining investment over the Base Case is projected to cost an additional $3.83 billion, with $1.48 billion being used for debottlenecking projects and $1.96 billion used for new units. For the high priced crude case, the incremental capital investments for an ECA is $3.44 billion over the base case, with new units accounting for $2.49 billion while debottlenecking costs are $0.72 billion. For both of the crude oil price cases, refinery investments represent a marginal increase of about 2 percent over the corresponding total base case investments required in 2020. Additionally, the majority of these ECA investments occur in the U.S./Canada refining regions, though smaller amounts also occur in other world regions. In addition to increased oil price estimates, the updated model accounts for increases in natural gas costs, capital costs for refinery upgrades, and product distribution costs.

5.1.3 Results of Fuel Cost Study

5.1.3.1 Incremental Refinery Capital Investments Associated with Desulfurization

5.1.3.1.1 General Overview

The primary refining cost of desulfurization is associated with converting IFO bunker oil into a distillate fuel with a DMA specification. The other significant refining costs are those related to desulfurizing distillate stocks. The bulk of the refinery investments occur in regions located outside of the U.S. and Canada, because capital investments in these regions are approximately 9 and 23 percent of the overall capital for the reference and high priced crude cases, respectively. Table 5.1-1 summarizes the overall capital investments made for both conversion of IFO bunker oil into distillate as well as desulfurization in refineries in the various U.S. regions (East Coast, West Coast and Gulf Coast) and overseas. These cost estimates are based on the WORLD modeling results.

Table 5.1-1 Incremental Refinery Capital Investment Made in 2020 (2006 dollars)

	REFINERY INVESTMENTS ($ BILLION)					
	Base Case $52/bbl Crude	**NA ECA $52/bbl Crude**	**Delta**	**Base Case $88/bbl Crude**	**NA ECA $88/bbl Crude**	**Delta**
USEC	1.4	1.2	-0.2	1.0	0.9	-0.1
USGCCE	14.5	14.8	0.3	26.2	27.3	1.2
USWCCW	1.4	1.6	0.2	1.4	1.5	0.2
Refinery Investments Total USA+Canada	17.3	17.6	0.3	28.6	29.8	1.3
Refinery Investments Total Other Regions	85.2	88.1	2.9	110.5	115.0	4.4
Total World	102.5	105.7	3.2	139.1	144.8	5.7
Type of Modification						
Debottleneck	0.7	0.7	0.0	1.4	1.4	0.0
Major New Units	97.8	100.8	3.0	132.1	138.0	6.0
Total World	102.5	105.7	3.2	139.1	144.8	5.7

Note: USEC is United States East Coast, USGCCE is United States Gulf Coast and Eastern Canada, USWCCW is United States West Coast and Western Canada, $Bn is Billion U.S. Dollars. The results presented are investments made in 2020 to add new refinery processing capacity to what exists in the 2008 base case plus known projects.

Refinery investments in North America, Greater Caribbean and South American regions account for greater than half of all investments for the reference case, while investments made in China and Middle Eastern Gulf regions account for close to 40 percent of remaining investments. This accounts for greater than 90 percent of investments for the reference case. For the high priced ECA case, investments in U.S., Canada, Greater Caribbean and South American refiner regions again account for greater than half of all investments made, while European north and China regions account for greater than 44 percent of the remaining investments. Table 5.1-2 summarizes overall incremental investments made in all world refining regions for the reference and high priced ECA case.

Table 5.1-2 World Region Refining Investments for ECA Made in 2020

	REFERENCE CASE		HIGH PRICED CASE	
	Capital, $ Billion	% of Capital	Capital, $ Billion	% of Capital
USEC	-0.167	-5.2%	-0.095	-1.7%
USGICE	0.277	8.7%	1.159	20.3%
USWCCW	0.176	5.5%	0.224	3.9%
GrtCAR	0.253	7.9%	0.828	14.5%
SthAM	0.810	25.4%	0.870	15.3%
AfWest	0.004	0.1%	0.002	0.0%
AfN-EM	0.143	4.5%	-0.006	-0.1%
Af-E-S	0.007	0.2%	0.006	0.1%
EUR-No	0.011	0.4%	1.239	21.7%
EUR-So	-0.006	-0.2%	-0.035	-0.6%
EUR-Ea	0.021	0.7%	-0.014	-0.2%
CaspRg	0.157	4.9%	-0.001	0.0%
RusFSU	0.185	5.8%	0.036	0.6%
MEGulf	0.754	23.6%	0.119	2.1%
PacInd	-0.115	-3.6%	0.069	1.2%
PacHi	0.177	5.5%	0.000	0.0%
China	0.490	15.3%	1.305	22.9%
RoAsia	0.018	0.6%	-0.002	0.0%
Total	3.20	100.0%	5.70	100.0%

Note: USEC = US East Coast, USGICE= US Gulf Coast, Interior & Canada East, USWCCW= US West Coast & Canada West, GrtCAR= Greater Caribbean, SthAM= South America, AfWest=African West, AFN- EM= North Africa/Eastern Mediterranean, AF-E-S=Africa East and South, Eur-No=Europe North, EUR-So= Europe South, EUR-EA= Europe East, CaspRg= Caspian Region, RusFSU= Russia & Other Former Soviet Union, MEGulf= Middle East Gulf, Pac Ind= Pacific Industrialized, PacHi= Pacific High Growth / Industrialising, RoAsia= Rest of Asia

5.1.3.1.2 Processing of Residual Stocks

IFO bunker grades are primarily comprised of residual stocks, such as Vacuum Residuals, Atmospheric Residuals, Visbreaker Residuals, and Fluidized Catalytic Cracking (FCC) clarified oil. These fuels also contain distillates that are added as cutter stocks, such as Light Cycle Oil (LCO), Vacuum Gas Oils (VGO), and kerosenes. As such, only the residual fuel blendstocks in IFO bunkers would need to be replaced or converted into distillate volumes to provide for additional lower sulfur distillate marine fuel. For converting residuals to distillates, refiners use two process technologies: Coking Units (Cokers) and Residual Hydrocrackers.

Coking units are used to convert the poorer quality residual feedstocks in IFO bunkers, such as Vacuum residuals. The coking units crack these residuals into distillates, using heat and residence time to make the conversion. The process produces petroleum coke and off gas as byproducts. Residual hydrocrackers are used to convert low and medium sulfur residual streams into distillates. Residual hydrocracking uses fluidized catalyst, heat and hydrogen to catalytically convert residual feedstocks into distillates and other light fuel products. The hydrocracking process upgrades low value residual stocks into high value distillate transportation fuels consuming large amounts of hydrogen.

For processing of residual blendstocks, vacuum tower distillation capacity is added to extract gas oils blendstocks that exist in residuals fuels used in current IFO bunker grades. The extracted gas oils are further processed in either distillate hydrotreaters or gas oil hydrocrackers to produce a distillate fuel that would meet a 0.1 percent fuel sulfur limit. The use of additional vacuum towers capacity minimizes the volume of residual stocks which lowers processing costs, as less volume of fuel is processed in high cost residual coking and residual hydrocracker processes.

5.1.3.1.3 Distillate Stocks Processing

Conventional distillate hydrotreating technology is used to lower the sulfur levels of high sulfur distillate stocks. This technology removes sulfur compounds from distillate stocks using catalyst, heat and hydrogen. Since the ECA sulfur standard is 0.1 percent, conventional distillate hydrotreating would likely be the technology chosen by refiners to make this distillate, rather than the ultra lower sulfur technology that is used to remove sulfur to levels below 15 ppm (0.0015 percent). Conventional distillate hydrotreating refers to the design and conditions in the process, such as catalyst type, catalyst volume, reactor pressure, feed and reactor flow scheme used to lower sulfur levels to 0.05 percent or higher.

Although the cutter stocks in IFO bunkers are distillate fuels, they would need to be desulfurized because the 0.1 percent sulfur limit for the ECA is lower than the nominal sulfur levels for these blendstocks under the "business as usual" projections. The sulfur levels of distillate used directly as bunker fuel (MDO and MGO), are greater than 1,000 ppm, and thus would also need to be treated. Therefore, in addition to converting residuals to distillate fuels, existing distillates used as bunker fuel in MDO, MGO and IFO would also need to be hydrotreated. More distillate hydrotreating capacity would be required to lower the sulfur content of incremental distillate produced from cokers and residual hydrocrackers that do not meet lower sulfur marine fuel standards.

For distillate stocks that are highly aromatic and high in sulfur, the use of technology for hydrocracking lower sulfur gas oil is used to convert these blendstocks into No 2. grade diesel streams. Gas oil hydrocracking is a high volume gain process which produces diesel blendstocks that typically meet ECA sulfur standards, eliminating the need for further processing in hydrotreaters.

5.1.3.1.4 Supportive Processes

The increase in hydrotreating and hydrocracking requires new hydrogen and sulfur plant capacity. Extra hydrogen is required to react with and remove sulfur compounds in refinery hydrotreating process. It is also needed to improve the hydrogen to carbon ratio of products made from converting IFO blend components to distillates, via processing in cokers and hydrocrackers.

5.1.3.2 Capacity and Throughput Changes for the Reference Case

The WORLD model used a total of 140 thousand barrels per stream day (KBPSD) of coking capacity to convert residual stocks to distillates. Of this amount, 110 KBPSD is existing spare or "slack" capacity available in U.S. and Canada refiner regions. This capacity is available based on projections that refiners add excess coking capacity in the base case. The remaining balance of coking capacity, or 30 KBPSD, is new capacity added to refiner regions outside of United States and Canada. In addition to utilizing more coking capacity, the WORLD model also increased residual hydrocracking capacity by 50 KBPSD to convert residual stocks into distillates. These hydrocrackers were added to refiner regions located outside of United States and Canada. Overall, considering the use of cokers and residual hydrocrackers, the total refiner process capacity is 190 KBPSD for residual stocks processing, mirroring the amount needed to process the residual volumes contained in IFO180 and IFO 380 bunker grades. To remove any gas oils in residual blendstocks such as atmospheric and vacuum tower residuals, the model utilized 60 KBPSD of existing vacuum tower capacity, 50 KBPSD in U.S. and Canada and 10 KBPSD in other refiner regions.

Crude throughput is increased by 54 KBPSD, primarily to account for increased energy usage in refinery processes such as hydro crackers and hydrotreaters. Crude throughput is also increased to offset liquid volume loss from residual stocks that are converted to petroleum coke in coking units. Table 5.1-3 summarizes overall crude and non crude throughputs for the base and ECA cases in units of million barrels per stream day (MMBPD).

Table 5.1-3 Refiner Crude and Non Crude Throughputs

		REFERENCE BASE CASE	REFERENCE ECA CASE	DELTA	HIGH BASE CASE	HIGH ECA CASE	DELTA
Crude Throughput	MMBPD	86.7	86.7	0.1	75.6	75.6	0.0
Non Crude Supply							
NGL ETHANE	MMBPD	1.7	1.7	0.0	1.7	1.7	0.0
NGLs C3+	MMBPD	6.3	6.3	0.0	6.1	6.1	0.0
PETCHEM RETURNS	MMBPD	1.0	1.0	0.0	0.8	0.8	0.0

BIOMASS	MMBPD	1.5	1.5	0.0	3.0	3.0	0.0
METHANOL (EX NGS)	MMBPD	0.1	0.1	0.0	0.1	0.1	0.0
GTL LIQUIDS (EX NGS)	MMBPD	0.3	0.3	0.0	0.6	0.6	0.0
CTL LIQUIDS (EX COAL)	MMBPD	0.5	0.5	0.0	0.8	0.8	0.0
HYDROGEN (EX NGS)	MMBPD	1.0	1.0	0.0	0.8	0.9	0.1
Total Non Crude Supply	MMBPD	12.3	12.3	0.0	14.0	14.0	0.0
TOTAL Supply	MMBPD	99.3	99.4	0.1	90.2	90.3	0.1

The model added 70 KBPSD of new ultra lower sulfur gas oil hydrocracking capacity in refiner regions outside of the U.S. and Canada. The distillate produced from these units has a sulfur content low enough to meet ECA standards and therefore does not require further processing in hydrotreaters. The model also reduced throughput by 40 KBPSD in existing base case capacity for Conventional Gas Oil Hydrocrackers located in U.S. and Canada refiner regions.

The model added 160 KBPSD of new conventional distillate hydrotreating capacity, 140 KBPSD to U.S. and Canada refiner regions and 20 KBPSD in refining regions in other areas of the world. In addition to new units, the model used 150 KBPSD of “slack” distillate conventional hydrotreating capacity, 90 KBPSD of this located in U.S. and Canada and 60 KBPSD in other world refiner regions. Considering this, the total net use of conventional distillate hydrotreating for the reference case is 310 KBPSD above the base case, mirroring incremental demand of lower sulfur distillate for ECA. The model used 70 KBPSD of slack capacity for vacuum gas oil/residual hydrotreating in addition to distillate hydrotreating. Of this amount, 40 KBPSD is in U.S. and Canada and 30 KBPSD in other world refiner regions.

The increased hydrotreating and hydrocracking capacity requires new hydrogen and sulfur plant capacity and was added to refiner regions that use more distillate hydrotreating and hydrocracking. Other minor refinery process modifications were required by the model in 2020, although these were not substantial (see Table 5.1-4).

Table 5.1-4 Refinery Secondary Processing Capacity Additions in 2020 Reference Case (Million barrels per stream day)

	USE OF BASE CAPACITY			NEW CAPACITY			**BASE PLUS NEW CAPACITY**		
	US/CAN	Rest of World	Total	US/CAN	Rest of World	Total	US/CAN	Rest of World	Total
Total Additions Over Base	0.00	0.05	0.05	0.00	0.05	0.05	0.00	0.05	0.05
Total Crude Capacity Used 2020	0.02	0.04	0.05	0.02	0.04	0.05	0.017	0.037	0.054
Vacuum Distillation	0.05	0.01	0.06	0.00	(0.02)	(0.02)	0.05	(0.01)	0.04
Coking	0.11	0.00	0.12	0.00	0.02	0.02	0.11	0.03	0.14
Catalytic Cracking	(0.07)	0.01	(0.06)	0.00	(0.01)	(0.01)	(0.07)	0.00	(0.07)
Hydro-Cracking (TOTAL)	(0.04)	0.00	(0.04)	0.00	0.12	0.12	(0.04)	0.12	0.08
- Gasoil Conventional	(0.04)	0.00	(0.04)	0.00	0.00	0.00	(0.04)	0.00	(0.04)
- Gasoil ULS	0.00	0.00	0.00	0.00	0.07	0.07	0.00	0.07	0.07
- Resid LS	0.00	0.00	0.00	0.00	0.01	0.01	0.00	0.01	0.01

- Resid MS	0.00	0.00	0.00	0.00	0.04	0.04	0.00	0.04	0.04
Catalytic Reforming with Revamp	0.01	0.00	0.02	0.00	0.07	0.07	0.01	0.07	0.08
Hydrotreating (Total)	0.13	0.08	0.21	0.11	0.05	0.17	0.24	0.14	0.37
- Gasoline – ULS	0.00	(0.00)	(0.00)	(0.03)	0.03	(0.00)	(0.03)	0.02	(0.01)
Distillate -New Conv/LS	0.09	0.06	0.15	0.14	0.02	0.16	0.23	0.08	0.31
- VGO/Resid	0.04	0.03	0.06	0.00	0.00	0.00	0.04	0.03	0.07
Hydrogen, (MMSCFD)	0	70	70	8	211	218	8	280	288
Sulfur Plant, (TPD)	500	500	1000	10	130	140	510	630	1140

While coking and hydrocracking (residual and gas oil) processes primarily produce distillates, to a lesser extent, some low octane gasoline blendstocks are also manufactured, requiring refiners to install additional catalytic reforming unit capacity. As such, in the U.S. and Canada regions approximately 10 KPBSD of existing spare catalytic reforming capacity is used while approximately 70 BPSD of new catalytic reforming capacity is added to other WORLD refiner regions that added cokers and hydrocrackers.

5.1.3.3 Capacity and Throughput Changes for the High Price Crude Oil Case

For the high priced case, the high cost of crude and high capital costs for processing units push the model to reduce installation of new processing units. The price of natural gas is also reduced relative to the price of crude which induces the model to use more natural gas and reduce the use of crude. Under these conditions, the model uses less crude, more natural gas and installs less capital for refinery processing units. As a result, the model favors the use of more hydrocracking processing which adds hydrogen (made from natural gas) to residual and gas oils, producing lower sulfur distillates stocks that do not require further processing in hydrotreaters. The model also uses more synthetic crudes and less heavy sour crudes, which reduce the amounts of residual stocks that need upgrading.

Crude throughput is increased by 29 KBPSD, which is less than the reference case, as the model preferentially uses natural gas over crude and reduces the use of cokers and hydrotreating. Table 5.1-5 shows crude and non crude inputs for the high priced case.

The WORLD model used a total of 80 KBPSD of "slack" coking capacity to convert residual stocks to distillates. Of this amount, 70 KBPSD was used in the U.S. and Canada regions and 10 KBPSD in regions in other areas of the world. The model also added 80 KBPSD of new low and medium sulfur residual hydrocracking capacity to convert residual stocks into distillates—20 KBPSD in the U.S. and Canada and 60 KBPSD in other world refiner regions. Overall, considering the use of cokers and residual hydrocrackers, the total refiner process capacity for residual stocks processing for use in the ECA is 160 KBPSD for the high priced case.

To extract gas oils from residual blendstocks, the model utilized 90 KBPSD of existing vacuum tower capacity—80 KBPSD in the U.S. and Canada and 10 KBPSD on other refiner regions. In addition, the model added 120 KBPSD of new ultra lower sulfur gas oil hydrocracking capacity in refiner regions outside of the U.S. and Canada. The distillate fuel produced from these units meet ECA sulfur standards. The model also used 30 KBPSD of slack capacity in the U.S. and Canada refiner regions for hydrocracking of conventional gas oil.

The model added 40 KBPSD of new conventional distillate hydrotreating capacity to the U.S. and Canada refiner regions and 20 KBPSD of new capacity to refining regions in other areas of the world. While the model also used 40 KBPSD of "slack" conventional distillate hydrotreating capacity in the U.S. and Canada, other world refiner regions decreased use of base case or slack capacity by 80 KBPSD. Considering the use of the new and slack capacity, a total net use of capacity is 20 KBPSD of conventional distillate hydrotreating capacity. The model also used 60 KBPSD of existing slack capacity for vacuum gas oil/residual distillate hydrotreaters, with 20 KBPSD used in the U.S. and Canada refiner regions and 40 KBPSD in other world refining regions.

The use of additional hydrocracking and hydrotreater capacity requires installation of new hydrogen plant capacity. New sulfur plant capacity is required in refiner regions to process the offgas produced from incremental use of hydro cracking and hydrotreating (see Table 5.1-5 below).

Table 5.1-5 Refinery Secondary Processing Capacity Additions in 2020 High Priced Case (Million barrels per stream day)

	USE OF BASE CAPACITY			NEW CAPACITY			BASE PLUS NEW CAPACITY		
	US/CAN	Rest of World	Total	US/CAN	Rest of World	Total	US/CAN	Rest of World	Total
Total Additions Over Base Case	0.00	(0.05)	(0.05)	0.00	(0.05)	(0.05)	0.00	(0.05)	(0.05)
Total Crude Capacity Used in 2020	0.05	(0.02)	0.03	0.05	(0.02)	0.03	0.054	(0.024)	0.029
Vacuum Distillation	0.08	0.10	0.18	0.00	0.00	0.00	0.08	0.10	0.18
Coking	0.07	0.01	0.08	0.00	(0.00)	(0.00)	0.07	0.00	0.08
Catalytic Cracking	(0.03)	(0.05)	(0.09)	0.00	0.00	0.00	(0.03)	(0.05)	(0.09)
Hydro-Cracking (Total)	0.03	0.00	0.03	0.02	0.18	0.20	0.05	0.18	0.22
- Gasoil Conventional	0.03	0.00	0.03	0.00	0.00	0.00	0.03	0.00	0.03
- Gasoil ULS	0.00	0.00	0.00	0.00	0.12	0.12	0.00	0.12	0.12
- Resid LS	0.00	0.00	0.00	0.02	0.03	0.05	0.02	0.03	0.05
- Resid MS	0.00	0.00	0.00	0.00	0.03	0.03	0.00	0.03	0.03
Catalytic Reforming with Revamp	0.00	0.02	0.02	(0.05)	0.02	(0.03)	(0.05)	0.04	(0.00)
Hydrotreating (Total)	0.06	(0.04)	0.02	0.04	0.02	0.06	0.11	(0.03)	0.08
- Gasoline – ULS	0.00	0.00	0.00	0.00	(0.01)	(0.01)	0.00	(0.01)	(0.01)
Distillate -New Conv/LS	0.04	(0.08)	(0.03)	0.04	0.02	0.06	0.08	(0.06)	0.02
- VGO/Resid	0.02	0.03	0.05	0.00	0.00	0.00	0.02	0.04	0.06
Hydrogen, (MMSCFD)	0	0	0	243	325	568	243	325	568
Sulfur Plant, (TPD)	580	300	880	0	120	120	580	420	1000

5.1.3.4 Overall Increases Due to Fuel Switching and Desulfurization

Global fuel use in 2020 by international shipping is projected to be 500 million tonnes/yr. The main energy content effects of bunker grade shifts were captured in the WORLD modeling by altering the volume demand and, at the same time, consistency was maintained between the bunker demand figures in tonnes and in barrels. The result was that partial or total conversion of IFO to distillate was projected to lead to a reduction in the total global <u>tonnes</u> of bunker fuel required but also led to a projected increase in the <u>barrels</u> required. These effects are evident in the WORLD case results. Based on the WORLD modeling, the volume of marine fuel affected by an ECA encompassing the U.S.[C] and Canadian EEZs would be about 4 percent of total world residual volume. As would be expected, since the shift in fuel volumes on a world scale is relatively small, the WORLD model predicts the overall global impact of an ECA to also be small.

There are two main components to projected increased marine fuel cost associated with an ECA. The first component results from the shifting of operation on residual fuel to operation on higher cost distillate fuel. This is the dominant cost component. The WORLD model computed costs based on a split between the costs of residual and distillate fuels. However, there is a small cost associated with desulfurizing the distillate to meet the 0.1 percent fuel sulfur standard in the ECA. Based on the WORLD modeling, the average increase in costs associated with switching from marine residual to distillate will be $145 per tonne.[D] This is the cost increase that will be borne by the shipping companies purchasing the fuel. Of this amount, $6 per tonne is the cost increase associated with distillate desulfurization. In other words, we estimate a cost increase of $6/tonne for distillate fuel used in an ECA.

The above cost estimates are based on EIA's "reference case" projections for crude oil price in 2020. We also performed a sensitivity analysis using EIA's "high price" scenario. Under this scenario, the increase in fuel costs for switching from residual to distillate fuel is $237 per tonne. The associated increase in distillate fuel cost is $7 per tonne.

Table 5.1-6 summarizes the reference and high price fuel cost estimates with and without an ECA. In the baseline case, fuel volumes for operation are 18% marine gas oil (MGO), 7% marine diesel oil (MDO), and 75% IFO. In the proposed ECA, all fuel volumes are modeled as MGO.

[C] For the contiguous U.S. and southeastern Alaska.

[D] Note that distillate fuel has a higher energy content, on a per tonne basis, than residual fuel. As such, there is an offsetting cost savings, on a per tonne basis, for switching to distillate fuel. Based on a 5 percent higher energy content for distillate, the net equivalent cost increase is estimated as $123 for each tonne of residual fuel that is being replaced by distillate fuel ($200/tonne for the high price case).

Table 5.1-6: Estimated Marine Fuel Costs

FUEL	UNITS	REFERENCE CASE		HIGH PRICE CASE	
		Baseline	ECA	Baseline	ECA
MGO	$/bbl	$ 61.75	$ 62.23	$ 102.70	$ 103.03
	$/tonne	$ 464	$ 468	$ 772	$ 775
MDO	$/bbl	$ 61.89	$ 62.95	$ 102.38	$ 103.70
	$/tonne	$ 458	$ 466	$ 757	$ 767
IFO	$/bbl	$ 49.87	$ 49.63	$ 83.14	$ 82.52
	$/tonne	$ 322	$ 321	$ 538	$ 534

5.2 Engine and Vessel Costs

This section presents the analysis of the potential cost impacts that the proposed ECA may have on new engines and vessels in the year 2020. To assess the potential cost impacts we must understand: the makeup of the fleet of ships expected to visit the U.S. when these requirements go into effect, the emission reduction technologies expected to be used, and the cost of these technologies. The total engine and vessel costs associated with the proposed ECA are based on a hardware cost per unit value applied to the number of affected vessels, and include operational costs. This section discusses an overview of the methodology used to develop a fleet of vessels expected to visit the U.S. portion of the proposed ECA, and presents the methodology used to develop the hardware and operational costs.

5.2.1 Overview

There are a number of technologies available or expected to be available to meet Tier III NO_X standards and to accommodate the use of lower sulfur fuel. We expect that each manufacturer will evaluate all possible technology avenues to determine how to best balance their respective costs while ensuring compliance; however, this analysis makes certain assumptions regarding how manufacturers will comply with the new emission and fuel standards. We expect that selective catalytic reduction (SCR) is the emission control technology most likely to be used to meet Tier III NO_X standards in the proposed ECA; therefore, this cost analysis is based on the use of SCR. With respect to fuel sulfur controls, we expect that switching to lower sulfur fuel is the most likely method of control to meet the fuel sulfur requirements when operating in the proposed ECA; therefore, this cost analysis is also based on switching to the use of lower sulfur fuel.

While fuel sulfur standards will take effect in 2015 and Tier III NO_X standards will take effect in 2016, this cost analysis only presents the hardware and operating costs that are expected to be incurred in 2020. In order to present the costs associated with the proposed ECA in 2020, the hardware costs are only applied to new vessels in 2020 expected to visit U.S. ports, while operating costs apply to all ships operating in the U.S. portion of the proposed ECA in 2020. The cost estimates presented here assume that all of the hardware costs for new ships in 2020 are due exclusively to this proposed ECA, and do not include an adjustment accounting for the potential existence of other ECAs that these ships may visit which would also require Tier III NO_X controls and appropriate fuel sulfur controls. The operational costs described in this section include those incurred in 2020 within the U.S. portion of the proposed ECA as a result of the use of urea on ships built as of 2016 equipped with SCR, and the differential costs associated with the use of lower sulfur fuel.

5.2.2 Methodology

To project future costs, we needed to first develop estimates of the number of ships that may visit the proposed ECA in 2020. To develop a future fleet, an approach similar to that used to estimate the emissions inventory (see Chapter 2) was used here. Specifically, the same inputs were used to develop a fleet of ships by ship type and engine type that may be expected to visit U.S. ports in 2020. Next, we needed to develop the estimated technology hardware costs, and sought input from the regulated community regarding the expected future costs of applying the emission control technologies associated with the proposed ECA. The U.S. Government contracted with ICF International to research the fixed and variable costs associated with the technologies expected to be used to meet Tier III NO_X and fuel sulfur standards.[9] To assess the cost of these new technologies, we developed a series of 'typical' engines with varying sizes and characteristics (e.g. stroke, number of cylinders, etc.) that the technologies would be applied to for the purposes of performing the cost research. The resulting cost estimates of applying different technologies to these 'typical' engines formed the basis for this cost analysis; Table 5.2-1 lists these engine configurations.

Table 5.2-1 Average Engine Characteristics Used in this Study

ENGINE TYPE	MEDIUM-SPEED			LOW-SPEED		
Engine Power (kW)	4,500	9,500	18,000	8,500	15,000	48,000
Cylinders	9	12	16	6	8	12
Liters/cylinder	35	65	95	380	650	1400
Engine Speed (rpm)	650	550	500	130	110	100
BSFC (g/kWh)	210			195		

After initial cost estimates were developed, ICF provided surveys to several engine and emission control technology manufacturers to determine the reasonableness of the approach and cost estimates. Input received from those surveyed was incorporated into the final cost estimates used in this analysis. The resulting costs for the 'typical' engines were plotted and a curve-fit was used to determine an equation to estimate the dollar-per-kilowatt ($/kW) cost for each technology. The hardware costs per vessel were based on average vessel characteristics (e.g. engine type and propulsion power) determined for various ship types. The per vessel costs were used with the estimated number of new vessels in 2020 expected to visit U.S. ports to evaluate the total hardware costs associated with the U.S. portion of the proposed ECA. The total operational costs were determined from the differential fuel cost estimates presented in Section 5.1 and the regional fuel consumption values presented in Chapter 2. For vessels equipped with SCR, urea consumption is expected to be 7.5 percent of the fuel consumption.

Operating costs per vessel vary depending on what year the vessel was built, for example, in 2020, vessels built as of 2016 will incur operating costs associated with the use of urea necessary when using SCR as a Tier III NO_X emission control technology, while vessels built prior to 2016 will only incur operating costs associated with the differential cost of using of lower sulfur fuel. To develop the costs associated with the proposed ECA in 2020, an approximation of the number of ships by age that may visit the proposed ECA in 2020 had to be constructed. To develop this future 2020 fleet, the data from ship calls to U.S. ports in the baseline year of 2002 were used to estimate how many ships would visit U.S. ports in 2020.[E,10]

5.2.2.1 2020 Fleet Development

The U.S. port data from 2002 used in the inventory port analysis and the regional growth rates presented in Chapter 2 were used to estimate how many ships by ship type and engine type may visit U.S. ports in the future. The ships that called on the U.S. in 2002 were cross referenced with Lloyd's database using their IMO numbers to determine the propulsion power, engine type, and ship type of each ship.[11] This allowed for all ships without Category 3 engines to be removed from the analysis. In order to separate slow speed engines (SSD) from medium speed engines (MSD) where that information was not explicitly available, 2-stroke engines were assumed to be SSD, and 4-stroke engines were assumed to be MSD. The research performed for this cost analysis differentiated between SSD and MSD engines, and separate $/kW values were developed for each of these engine types.

The ship type information gathered from this baseline data, for the purposes of both this analysis and the inventory, was categorized into one of the following ship types: Auto Carrier, Bulk Carrier, Container, General Cargo, Miscellaneous, Passenger, Refrigerated Cargo (Reefer), Roll-On Roll-Off (RoRo), and Tankers. The 2002 baseline fleet was also used to develop average ship characteristics shown in Table 5.2-2. These values were used to represent the characteristics of new (and future existing) vessels for the purposes of this cost analysis.

The 2002 port call data were sorted by IMO number to determine the total number of unique ships that visited all included U.S. ports in 2002. Table 5.2-3 shows the breakout by ship type of these approximately 6,700 ships. Next, in order to be consistent with the inventory analysis which presents growth rates by region, the original port call data was separated into the same regions used by the inventory (South Pacific (SP), North Pacific (NP), East Coast (EC), Gulf Coast (GC), Alaska East (AE), Alaska West (AW), Hawaii East (HE), and West Hawaii (HW)). This was done by matching each port-of-call entry in the original port call data file with the corresponding region containing that port as per the inventory analysis.[12] This resulted in a fleet of ships for each region, each with a unique IMO number as shown in Table 5.2-3.

[E] The 2002 U.S. ship call data used to determine the 2002 baseline fleet was also used to construct port inventories, as discussed in the Emissions Inventory Chapter. As such, this fleet includes the same ports and limitations as the inventory analysis (e.g. military vessels are excluded, as are ships powered by engines <30 L/cyl.)

Table 5.2-2 Average Ship Characteristics used in this Cost Analysis

SHIP TYPE	ENGINE SPEED	AVERAGE PROPULSION POWER (KW)	AVERAGE AUXILIARY POWER (KW)	SERVICE SPEED (KNOTS)	AVERAGE DWT
Auto Carrier	Slow Speed	11,000	3,000	19	17,000
	Medium Speed	9,600	2,600	17	13,000
Bulk Carrier	Slow Speed	8,400	1,900	15	47,000
	Medium Speed	6,300	1,400	14	27,000
	Steam Turbine	6,400	1,400	15	19,000
Container	Slow Speed	27,000	6,000	22	45,000
	Medium Speed	14,000	3,000	19	19,000
	Steam Turbine	21,000	4,700	21	30,000
General Cargo	Slow Speed	7,700	2,000	15	26,000
	Medium Speed	5,200	1,300	15	8,700
	Steam Turbine	18,000	4,600	21	23,000
Passenger	Slow Speed	24,000	6,600	210	6,200
	Medium Speed	24,000	6,600	20	6,200
	Steam Turbine	27,000	7,600	19	13,000
	Gas Turbine	44,000	12,000	24	12,000
Reefer	Slow Speed	10,000	4,200	20	11,000
	Medium Speed	7,400	3,000	18	7,600
RoRo	Slow Speed	16,000	4,000	18	30,000
	Medium Speed	8,600	2,200	16	8,400
	Gas Turbine	47,000	12,000	24	37,000
	Steam Turbine	22,000	5,800	25	19,000
Tanker	Slow Speed	9,800	2,100	15	61,000
	Medium Speed	6,700	1,400	15	27,000
	Gas Turbine	7,600	1,600	15	40,000
	Steam Turbine	21,000	4,400	18	59,000
Misc.	Slow Speed	4,700	1,300	14	8,800
	Medium Speed	9,400	2,500	13	6,000
	Steam Turbine	13,000	3,500	21	17,000

Some ships may have visited ports in more than one region which could result in an overestimate of the hardware costs (which are applied to each unique vessel) if the number of vessels in each region were grown, summed together and used for the total costs. To prevent over-counting of vessels visiting U.S. ports, a factor was developed (see Equation 1) to account for this overlap. The number of unique ships in each region (identified by unique IMO numbers) was summed together to produce a total number of "unique" ships visiting all regions, this value was reduced by the total number of actual unique ships that visited U.S. ports in 2002 (from the original baseline data) to provide a factor representing the original number of unique ships visiting U.S. ports. This factor was then applied to the vessel count in each region to provide a regional total that would coincide with the baseline total, and eliminate the over-counting of ships that had visited multiple regions.

Equation 1 Regional Fleet Overlap Reduction Factor Example

$$\frac{\#Unique\_Auto\_Carriers\_in\_Total\_Port\_Call\_Data}{\sum Unique\_Auto\_Carriers\_by\_\mathrm{Re}gion} = \%\_Actual\_Unique\_\mathrm{Re}gional\_Auto\_Carriers$$

For example, a total of 300 unique auto carriers visited all included U.S. ports in 2002, yet when looking at unique ships on a regional basis and totaling all regions, 650 auto carriers appeared to visit. This implied that only 46 percent of the regional auto carriers were "unique" and that the additional 350 auto carriers were ships that had visited multiple regions. Therefore, only 46 percent of all auto carriers within each regional fleet were assumed to be "unique." The growth rates were only applied to this corrected count of "unique" ships in each region to estimate the regional fleet makeup in future years.

Table 5.2-3 2002 Baseline Fleet of Ships and Regional Overlap Factor

SHIP TYPE	TOTAL UNIQUE SHIP VISITS TO U.S. PORTS IN 2020	REGIONAL UNIQUE SHIPS VISITING U.S. PORTS IN 2020	REGIONAL OVERLAP FACTOR
Auto Carrier	300	650	46%
Bulk	2,500	3,600	68%
Container	1,000	1,600	63%
Gen. Cargo	980	1,700	57%
Misc	24	50	49%
Pass	110	200	57%
Reefer	280	400	71%
RoRo	120	200	58%
Tanker	1,400	2,700	52%
Total	6,700	11,000	62%

Within each region, the ship types were further broken down by engine type. The unique ship fleet within each region was then grown by ship type and engine type using the appropriate growth rate to estimate the makeup of the future fleet in 2020. Table 5.2-4 shows the estimated 2020 fleet of ships expected to visit U.S. ports.

Table 5.2-4 Estimated 2020 Fleet by Ship Type and Engine Type

SHIP TYPE	ENGINE TYPE	NUMBER OF NEW VESSELS	NUMBER OF EXISTING VESSELS
Auto Carrier	SSD	45	570
	MSD	4	55
Bulk Carrier	SSD	440	5500
	MSD	8	110
	ST	3	21
Container	SSD	210	2600
	MSD	8	95
	ST	9	72

SHIP TYPE	ENGINE TYPE	NUMBER OF NEW VESSELS	NUMBER OF EXISTING VESSELS
General Cargo	SSD	100	1300
	MSD	57	95
	ST	0	3
Passenger	SSD	1	9
	MSD	8	110
	ST	1	5
	GT	1	8
Reefer	SSD	35	440
	MSD	6	80
RoRo	SSD	7	78
	MSD	3	38
	GT	0	3
	ST	0	2
Tanker	SSD	220	2700
	MSD	16	200
	GT	0	5
	ST	8	59
Misc.	SSD	0	1
	MSD	0	5
Total:		1,200	14,000

5.2.2.2 Existing Fleet That May Require Retrofit to Use Low Sulfur Fuel

Although most ships primarily operate on residual fuel, they typically carry some amount of distillate fuel as well. This distillate fuel is available for use in emergencies such as mechanical breakdown, off-spec bunker delivery, or prior to an extended engine shut-down to clear the residual fuel out of the heaters and piping. Switching to the use of lower sulfur distillate fuel is the compliance strategy assumed here to be used by both new and existing ships when the new fuel sulfur standards go into effect. To estimate the potential cost of this compliance strategy, we first evaluated the distillate storage capacity of the current existing fleet to estimate how many ships may require additional hardware to accommodate the use of lower sulfur fuel. We performed this analysis on the entire global fleet listed in Lloyd's database as of 2008. Of the nearly 43,000 vessels listed, approximately 20,000 vessels had provided Lloyd's with fuel tankage information, cruise speed, and propulsion engine power data. Using this information, we were able to estimate how far each vessel could travel on its existing distillate carrying capacity.

The cruise speed provided by Lloyd's was used to determine the vessel's maximum speed using Equation 2 while transit speed was assumed to be 12 knots, and maneuver speed 5.8 knots.[13] The load factor used at cruise speed was 83 percent; while both the transit and maneuver load factors were estimated by cubing the ratio their respective speeds to the ship's maximum speed. The same low load factors used in the inventory (for loads less than 20 percent) were used here to adjust brake specific fuel consumption (BSFC) because diesel engines are less efficient at low loads and the BSFC tends to increase. It was also assumed that ships spent a total of four hours per call in both transit and maneuver speeds. The fuel consumption values used here were the same as reported in the inventory section, 195 g/kWh for SSD, 210 g/kWh for MSD, and 305 g/kWh for steam and gas turbines. The fuel consumed by auxiliary engines was also taken into account and the same auxiliary power ratios used in the inventory analysis were used here to estimate the total installed auxiliary engine power, as were the auxiliary engine load factors appropriate for when the vessel is at cruise, transit, and maneuver speeds for each ship.

Equation 2: Maximum Speed

$$\frac{Lloyds\_speed}{0.94} * 0.83 = \max imum\_speed$$

In order to determine if the current distillate capacity of a particular ship was sufficient to call on a U.S. ECA without requiring additional hardware, we evaluated whether or not each ship could travel 1,140 nm, the distance between the Port of Los Angeles and the Port of Tacoma. This distance was selected because it represents one of the longer trips a ship could travel without stopping at another port, and should overestimate the number of vessels that would require such a modification. The amount of fuel a ship would consume calling on a port and travelling a total distance of 1,140 nm was determined using the methodology described above. The total fuel used in each mode (cruise, transit and maneuver) by both main and auxiliary engines was summed and compared to the total amount of distillate fuel carried onboard. This provided an estimate of the number of ships that had sufficient distillate capacity onboard, shown in Table 5.2-5. The resulting percentages of ships determined to require a retrofit were then applied to the number of existing ships in the 2015 fleet to estimate the total cost of this compliance strategy for existing ships. The same percentages were also applied to all new ships projected to be built in 2020 to determine the number of ships that may require additional hardware and to estimate the cost of this compliance strategy for new vessels in 2020.

Table 5.2-5 Ships that Can Travel 1,140 nm on Existing Distillate Carrying Capacity

SHIP TYPE	TOTAL # OF SHIPS	TOTAL # OF SHIPS THAT ONLY CARRY DISTILLATE	TOTAL # OF SHIPS THAT CARRY DISTILLATE + ANOTHER FUEL	SHIPS THAT CARRY DISTILLATE + ANOTHER FUEL THAT MAY NEED A MODIFICATION		TOTAL # OF SHIPS THAT CARRY NO DISTILLATE	% NO DISTILLATE	TOTAL OF ALL SHIPS THAT MAY NEED A MODIFICATION	
				#	%			#	%
General Cargo	4600	1900	2300	200	8.9%	370	8.2%	580	13%
Tanker	5900	740	4900	1600	33%	280	4.7%	1900	33%
Container	1900	45	1700	910	53%	140	7.3%	1000	55%
Bulk Cargo	3600	230	3000	1600	53%	400	11%	2000	55%
RoRo	510	70	380	30	7.6%	60	12%	90	18%
Auto Carrier	360	20	310	20	7.1%	40	10%	60	16%
Misc.	1600	1100	210	70	34%	210	14%	280	18%
Passenger	710	170	460	270	59%	85	12%	360	51%
Reefer	530	60	440	20	4.1%	25	4.8%	40	8.2%
Total	19,710	4,335	13,700	4,720	24%	1,610	8%	6,310	32%

5.2.3 Tier III NO_X Emission Reduction Technologies

The Selective Catalytic Reduction (SCR) process involves injecting a reagent, such as ammonia or urea, into an exhaust flow, upstream of a reactor, to reduce NO_X compounds into nitrogen and water. Main system components are: an SCR reactor, aqueous urea injection/dosing, and monitoring/control systems. The SCR system does require storage of urea solution onboard in a separate tank. In addition to SCR, it is expected that manufacturers will also use compound or two-stage turbocharging as well as electronic valving to enhance performance and emission reductions to meet Tier III NO_X standards. Engine modifications to meet Tier III emission levels may also include a higher percentage of common rail fuel injection coupled with two-stage turbocharging and electronic valving.

5.2.4 SO_X/PM Emission Reduction Technology

In addition to Tier III NO_X standards, the IMO ECA standards also include reductions in fuel sulfur limits that will result in reductions in SO_X and PM. While there are many existing ships that already have the capacity to operate on both heavy fuel oil and distillate fuel and have separate fuel tank systems to support each type of fuel, some ships may not have sufficient onboard storage capacity to accommodate temporary fuel switching to operate both main and auxiliary engines on lower sulfur fuel, since the minimum space practical is devoted to fuel and machinery to maximize cargo space. If additional capacity is required, installation and use of a fuel cooler, associated piping, and viscosity meters to the fuel treatment system may be required to ensure viscosity matches between the fuel and injection system. If a new or segregated tank is desired, ancillary equipment such as pumps, piping, vents, filling pipes, gauges, and access would be required, as well as tank testing.[14]

5.2.5 NO_X Emission Reduction Technology per Unit Hardware Costs

Tier III NO_X standards are approximately 80 percent lower than the existing Tier I NO_X standards set by the IMO. To meet these standards, it is expected that SCR will be used along with additional migration from either mechanically controlled mechanical fuel injection systems (MFI) or electronically controlled fuel injection systems (EFI) to common rail, and engine modifications. The methodology used here to estimate the capacity of the SCR systems is based on the power rating of the propulsion engines only. Auxiliary engine power represents about 20 percent of total installed power on a vessel; however, it would be unusual to operate both propulsion and auxiliary engines at 100 percent load. Typically, ships operate under full propulsion power only while at sea when the SCR is not operating; when nearing ports the auxiliary engine is operating at high loads while the propulsion engine is operating at very low loads. It is estimated that the remaining 20 percent of SSD engines (5 percent MFI and 15 percent EFI) that have not already been upgraded to common rail to meet global Tier II NOx standards will receive that upgrade for Tier III, and 40 percent of MSD (10 percent MFI and 30 percent EFI) will get common rail for Tier III as well. The fixed and variable costs of the six 'typical' engines developed for the migration to common rail from MFI are shown in Table 5.2-6.

Table 5.2-6 Fixed and Variable Costs for MFI to Common Rail Fuel Injection Systems

SPEED	**MEDIUM**	**MEDIUM**	**MEDIUM**	**LOW**	**LOW**	**LOW**
Engine Power (kW)	**4,500**	**9,500**	**18,000**	**8,500**	**15,000**	**48,000**
Cylinders	**9**	**12**	**16**	**6**	**8**	**12**
Liters/cylinder	**35**	**65**	**95**	**380**	**650**	**1400**
Engine Speed (rpm)	**650**	**550**	**500**	**130**	**110**	**100**
VARIABLE COSTS						
Component Costs						
Electronic Control Unit	$3,500	$3,500	$3,500	$5,000	$5,000	$5,000
Common Rail Accumulators (each)	$2,000	$2,000	$2,000	$2,000	$2,000	$2,000
Number of Accumulators	3	6	8	9	12	18
Low Pressure Pump	$2,000	$3,000	$4,000	$2,500	$3,500	$4,500
High Pressure Pump	$3,500	$4,500	$6,000	$4,500	$6,000	$8,000
Modified injectors (each)	$2,500	$2,500	$2,500	$3,500	$3,500	$3,500
Number of injectors	9	12	16	18	24	36
Wiring Harness	$2,500	$2,500	$2,500	$3,000	$3,000	$3,000
Total Component Cost	**$40,000**	**$55,500**	**$72,000**	**$96,000**	**$125,500**	**$182,500**
Assembly						
Labor (hours)	120	160	200	200	250	300
Cost ($23.85/hr)	$2,900	$3,800	$4,800	$4,800	$5,900	$7,100
Overhead @ 40%	$1,100	$1,500	$1,900	$1,900	$2,400	$2,900
Total Assembly Cost	**$4,000**	**$5,300**	**$6,700**	**$6,700**	**$8,300**	**$10,000**
Total Variable Cost	**$44,000**	**$60,800**	**$78,700**	**$102,700**	**$133,800**	**$192,500**
Markup @ 29%	$12,800	$17,700	$22,800	$29,800	$38,800	$55,800
Total Hardware RPE	**$56,800**	**$78,500**	**$101,500**	**$132,500**	**$172,600**	**$248,300**
FIXED COSTS						
R&D Costs (1 year R&D)	$688,000	$688,000	$688,000	$688,000	$688,000	$688,000
Retooling Costs	$1,000,000	$1,000,000	$1,000,000	$1,000,000	$1,000,000	$1,000,000
Marine Society Approval	$5,000	$5,000	$5,000	$5,000	$5,000	$5,000
Engines/yr.	40	40	40	40	40	40
Years to recover	5	5	5	5	5	5
Fixed cost/engine	**$8,500**	**$8,500**	**$8,500**	**$8,500**	**$8,500**	**$8,500**

The fixed and variable costs associated with the migration from EFI to common rail are shown in Table 5.2-7. A curve-fit to estimate the variable cost of each technology was then used to determine a $/kW equation applicable to other engine sizes and types, Figure 5-1 shows the curve-fit for MFI to common rail variable costs and Figure 5-2 shows the curve fit for EFI to common rail variable costs.

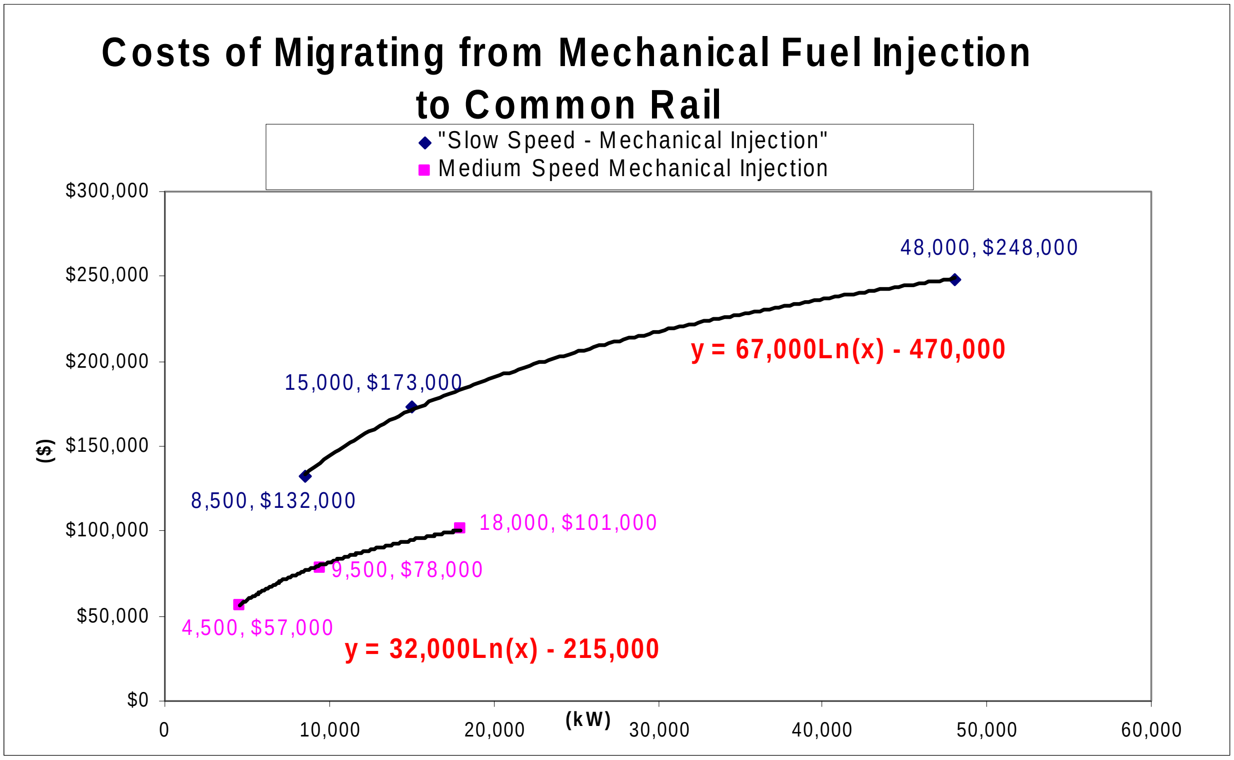

Figure 5-1 Variable Cost Curve-Fit for MFI to Common Rail Fuel Injection Systems

Table 5.2-7 Fixed and Variable Costs for EFI to Common Rail Fuel Injection Systems

SPEED	**MEDIUM**	**MEDIUM**	**MEDIUM**	**LOW**	**LOW**	**LOW**
Engine Power (kW)	**4,500**	**9,500**	**18,000**	**8,500**	**15,000**	**48,000**
Cylinders	**9**	**12**	**16**	**6**	**8**	**12**
Liters/cylinder	**35**	**65**	**95**	**380**	**650**	**1400**
Engine Speed (rpm)	**650**	**550**	**500**	**130**	**110**	**100**
Hardware Costs to the Manufacturer						
Component Costs						
Electronic Control Unit	$500	$500	$500	$500	$500	$500
Common Rail Accumulators (each)	$2,000	$2,000	$2,000	$2,000	$2,000	$2,000
Number of Accumulators	3	6	8	9	12	18
Low Pressure Pump	$1,000	$1,000	$1,000	$1,500	$1,500	$1,500
High Pressure Pump	$1,500	$1,500	$1,500	$2,000	$2,000	$2,000
Modified injectors (each)	$500	$500	$500	$750	$750	$750
Number of injectors	9	12	16	18	24	36
Wiring Harness	$500	$500	$500	$650	$650	$650
Total Component Cost	**$14,000**	**$21,500**	**$27,500**	**$36,150**	**$46,650**	**$67,650**
Assembly						
Labor (hours)	40	60	80	40	60	80
Cost ($23.85/hr)	$950	$1,430	$1,910	$950	$1,430	$1,910

Overhead @ 40%	$380	$570	$760	$380	$570	$760
Total Assembly Cost	**$1,330**	**$2,000**	**$2,670**	**$1,330**	**$2,000**	**$2,670**
Total Variable Cost	**$15,300**	**$23,500**	**$30,200**	**$37,500**	**$48,700**	**$70,300**
Markup @ 29%	$4,400	$6,800	$8,800	$10,900	$14,100	$20,400
Total Hardware RPE	**$19,700**	**$30,300**	**$39,000**	**$48,400**	**$62,800**	**$90,700**
FIXED COSTS						
R&D Costs (0.5 year R&D)	$344,000	$344,000	$344,000	$344,000	$344,000	$344,000
Retooling Costs	$500,000	$500,000	$500,000	$500,000	$500,000	$500,000
Marine Society Approval	$5,000	$5,000	$5,000	$5,000	$5,000	$5,000
Engines/yr.	40	40	40	40	40	40
Years to recover	5	5	5	5	5	5
FIXED COST/ENGINE	**$4,200**	**$4,200**	**$4,200**	**$4,200**	**$4,200**	**$4,200**

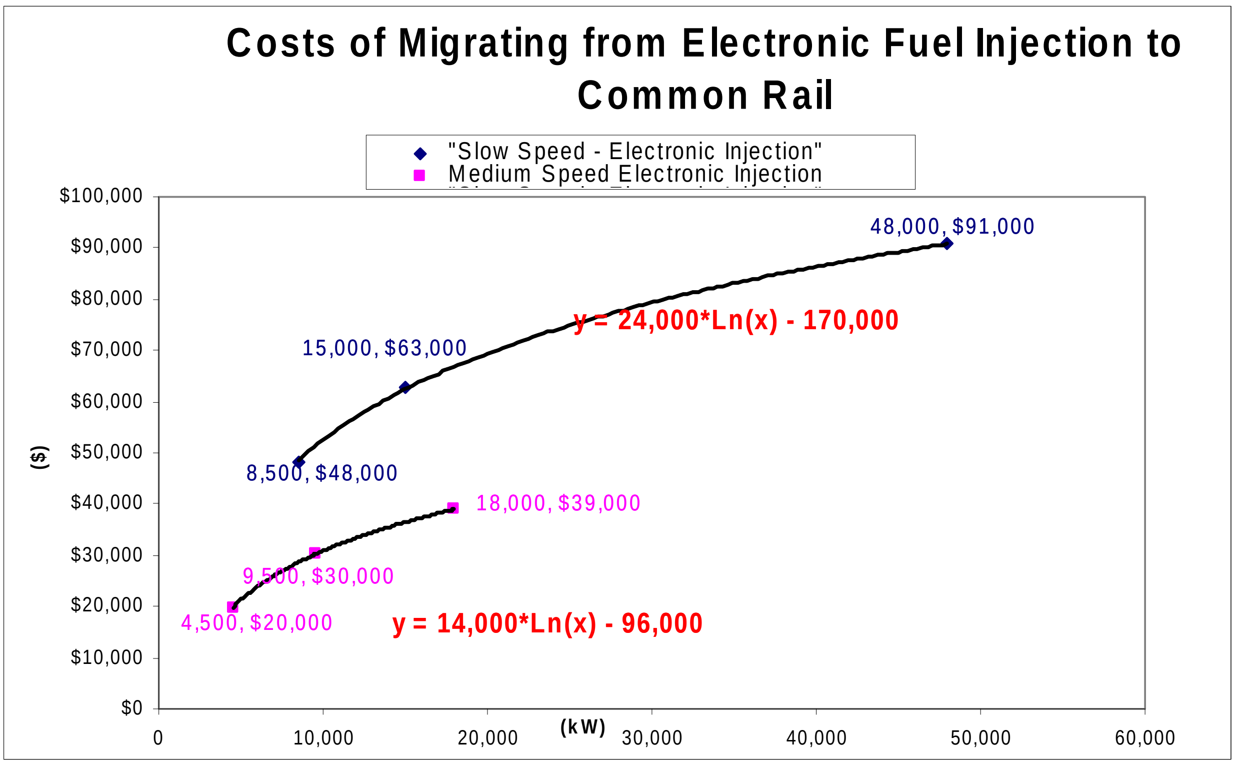

Figure 5-2 Cost Curve-Fit for EFI to Common Rail Fuel Injection Systems

The variable costs associated with the use of engine modifications for Tier III include the use of two stage turbochargers and electronic valve actuation, and are shown with the estimated fixed costs in Table 5.2-8, Figure 5-3 shows the variable cost curve-fit used to determine a $/kW equation applicable to other engine sizes and types. Table 5.2-9 shows the variable costs associated with the use of SCR, these costs include the urea tank, the reactor, dosage pump, urea injectors, piping, bypass valve, the acoustic horn, a cleaning probe and the control unit and wiring. Detailed costs for the urea tank are shown in Table 5.2-10 and are based on estimated storage of urea sufficient for up to 250 hours of normal operation of the SCR. It is envisioned that the urea tank is constructed of 304 stainless steel, 1 mm thick due to the corrosive nature of urea, at a cost of approximately $2,700 per metric tonne.[F] The cost of Tier III technology as presented here was developed using Tier II as a baseline. Figure 5-4 shows the shows the cost curve used to determine a $/kW equation applicable to other engine types and sizes. The total variable hardware costs of Tier III estimated here include the fuel injection changes, engine modifications, and SCR.

Table 5.2-8 Fixed and Variable Costs for Engine Modifications Associated with Tier III

SPEED	**MEDIUM**	**MEDIUM**	**MEDIUM**	**LOW**	**LOW**	**LOW**
Engine Power (kW)	**4,500**	**9,500**	**18,000**	**8,500**	**15,000**	**48,000**
Cylinders	**9**	**12**	**16**	**6**	**8**	**12**
Liters/cylinder	**35**	**65**	**95**	**380**	**650**	**1400**
Engine Speed (rpm)	**650**	**550**	**500**	**130**	**110**	**100**
Hardware Costs to the Manufacturer						
Component Costs						
2 Stage Turbochargers (Incremental)	$16,250	$20,900	$46,750	$28,000	$42,000	$61,000
Electronic Intake Valves (each)	$285	$285	$285			
Intake Valves per Cylinder	2	2	2			
Electronic Exhaust Valves (each)	$285	$285	$285	$425	$425	$425
Exhaust Valves per Cylinder	2	2	2	4	4	4
Controller	$3,750	$3,750	$3,750	$3,750	$3,750	$3,750
Wiring	$2,800	$2,800	$2,800	$2,800	$2,800	$2,800
Total Component Cost	**$33,000**	**$41,000**	**$72,000**	**$45,000**	**$62,000**	**$88,000**
Markup @ 29%	$10,000	$12,000	$21,000	$13,000	$18,000	$25,000
Total Hardware RPE	**$43,000**	**$53,000**	**$93,000**	**$58,000**	**$80,000**	**$113,000**
Fixed Costs						
R&D Costs (1 year R&D)	$688,000	$688,000	$688,000	$688,000	$688,000	$688,000
Retooling Costs	$1,000,000	$1,000,000	$1,000,000	$1,320,000	$1,320,000	$1,320,000
Marine Society Approval	$5,000	$5,000	$5,000	$5,000	$5,000	$5,000
Engines/yr.	40	40	40	40	40	40
Years to recover	5	5	5	5	5	5
Fixed cost/engine	**$8,500**	**$8,500**	**$8,500**	**$10,000**	**$10,000**	**$10,000**

[F] http://www.metalprices.com/FreeSite/metals/stainless_product/product.asp#Tables for 2006.

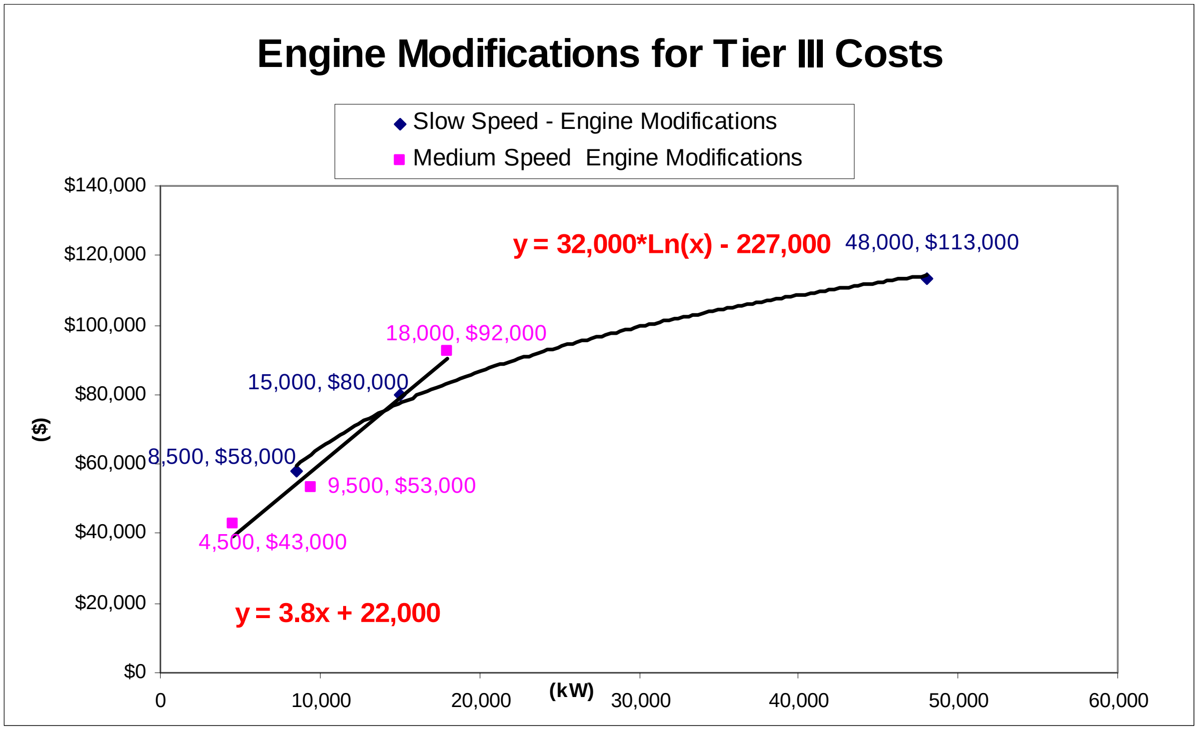

Figure 5-3 Variable Cost Curve-Fit for Engine Modifications Associated with Tier III

Table 5.2-9 Fixed and Variable Costs Associated with the Use of SCR

SPEED	**MEDIUM**	**MEDIUM**	**MEDIUM**	**LOW**	**LOW**	**LOW**
Engine Power (kW)	**4,500**	**9,500**	**18,000**	**8,500**	**15,000**	**48,000**
Cylinders	**9**	**12**	**16**	**6**	**8**	**12**
Liters/cylinder	**35**	**65**	**95**	**380**	**650**	**1400**
Engine Speed (rpm)	**650**	**550**	**500**	**130**	**110**	**100**
Hardware Costs to the Supplier						
Component Costs						
Aqueous Urea Tank	$1,200	$1,900	$2,800	$1,700	$2,400	$4,600
Reactor	$200,000	$295,000	$400,000	$345,000	$560,000	$1,400,000
Dosage Pump	$9,500	$11,300	$13,000	$11,300	$13,000	$15,000
Urea Injectors (each)	$2,400	$2,400	$2,400	$2,400	$2,400	$2,400
Number of Urea Injectors	3	6	8	12	16	24
Piping	$4,700	$5,600	$6,600	$5,600	$7,500	$9,500
Bypass Valve	$4,700	$5,600	$6,600	$5,600	$6,600	$7,500
Acoustic Horn	$9,500	$11,300	$13,000	$11,700	$14,000	$16,400
Cleaning Probe	$575	$575	$575	$700	$700	$700
Control Unit/Wiring	$14,000	$14,000	$14,000	$19,000	$19,000	$19,000
Total Component Cost	**$251,000**	**$360,000**	**$476,000**	**$429,000**	**$662,000**	**$1,530,000**
Assembly						
Labor (hours)	1000	1200	1500	1200	1600	2000

Cost ($23.85/hr)	$23,900	$28,600	$35,800	$28,600	$38,200	$47,700
Overhead @ 40%	$9,500	$11,400	$14,300	$11,400	$15,300	$19,100
Total Assembly Cost	**$33,400**	**$40,000**	**$50,100**	**$40,000**	**$53,500**	**$66,800**
Total Variable Cost	$284,800	$399,700	$525,800	$469,400	$715,000	$1,597,100
Markup @ 29%	$82,600	$115,900	$152,500	$136,100	$207,300	$463,200
Total Hardware RPE	**$367,400**	**$515,600**	**$678,300**	**$605,500**	**$922,300**	**$2,060,300**
Fixed Costs						
R&D Costs (1 year R&D)	$1,376,000	$1,376,000	$1,376,000	$1,376,000	$1,376,000	$1,376,000
Retooling Costs	$2,000,000	$2,000,000	$2,000,000	$2,000,000	$2,000,000	$2,000,000
Marine Society Approval	$5,000	$5,000	$5,000	$5,000	$5,000	$5,000
Engines/yr.	40	40	40	40	40	40
Years to recover	5	5	5	5	5	5
Fixed cost/engine	**$16,900**	**$16,900**	**$16,900**	**$16,900**	**$16,900**	**$16,900**

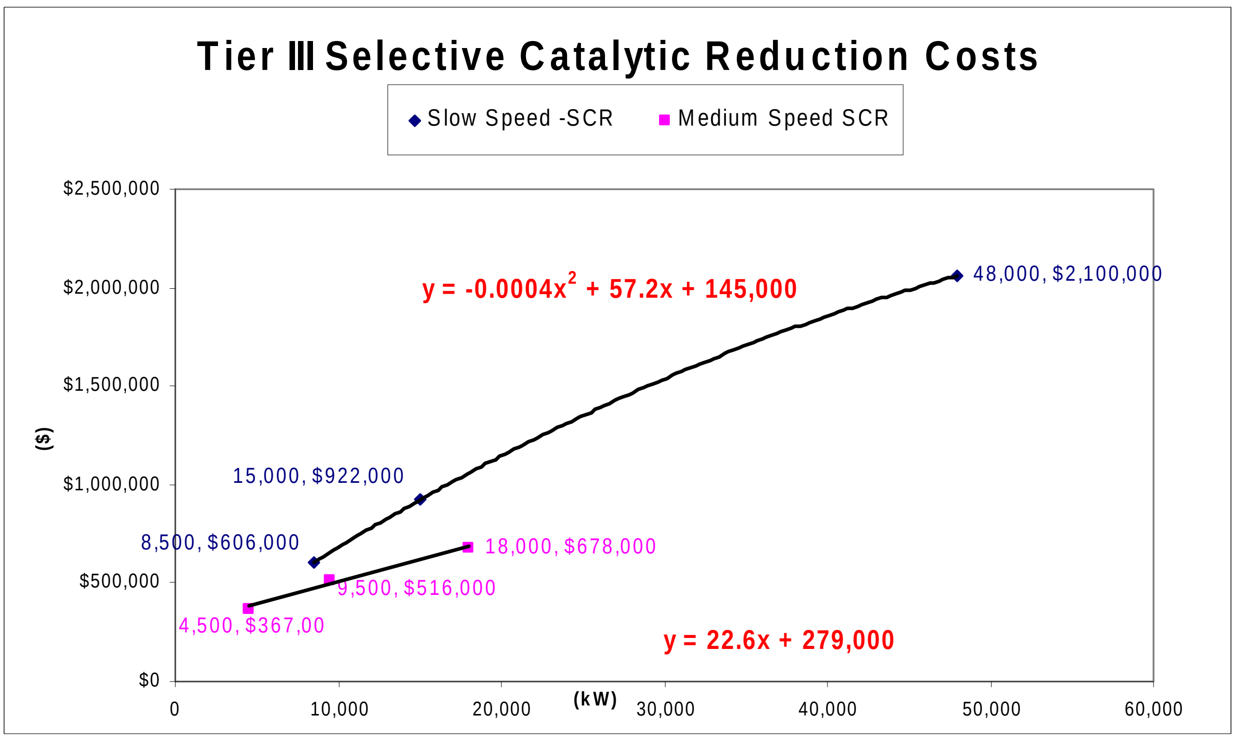

Figure 5-4 Variable Cost Curve-Fit for SCR Systems

Table 5.2-10 Detailed Urea Tank Variable Costs

SPEED	MEDIUM	MEDIUM	MEDIUM		LOW	LOW	LOW
Engine Power (kW)	4,500	9,500	18,000		8,500	15,000	48,000
Cylinders	9	12	16		6	8	12
Liters/cylinder	35	65	95		380	650	1400
Engine Speed (rpm)	650	550	500		130	110	100
Urea Tank Costs							
Urea Amount (kg)	12,910	27,255	51,642		22,645	39,961	127,875
Density (kg/m^3)	1,090	1,090	1,090		1,090	1,090	1,090
Tank Size (m^3)	14	30	57		21	37	117
Tank Material (m^3)	0.04	0.06	0.09		0.05	0.07	0.14
Tank Material Cost ($)	$758	$1,248	$1,909		$977	$1,426	$3,093
Assembly							
Labor (hours)	5	6	7		10	12	15
Cost ($/hr)	$119	$143	$167		$238	$286	$358
Overhead @ 40%	$48	$57	$67		$95	$114	$143
Total Assembly Cost	$167	$200	$234		$334	$401	$501
Total Variable Cost	$925	$1,448	$2,143		$1,310	$1,826	$3,594
Markup @ 29%	$268	$420	$621		$380	$530	$1,042
Total Hardware RPE	$1,194	$1,868	$2,765		$1,690	$2,356	$4,636

5.2.6 SO_X and PM Emission Reduction Technology per Unit Hardware Costs

As discussed above, this cost analysis is based on the use of switching to lower sulfur fuel to meet the ECA fuel sulfur standards when operating in the U.S portion of the proposed ECA. This section discusses the costs that may be incurred by some newly built ships if additional fuel tank equipment, beyond that installed on comparable new ships, is required to meet lower sulfur fuel standards in the proposed ECA. We estimate that nearly one-third of new vessels in 2020 may need additional equipment installed to accommodate additional lower sulfur fuel storage capacity. The size of the tank is dependent on the frequency with which the individual ship owner prefers to fill the lower sulfur fuel tank. The size of the tanks as estimated here will carry capacity sufficient for 250 hours of propulsion and auxiliary engine operation while within an ECA. Similar to the urea tank size estimation presented in this analysis, this is most likely an overestimate of the amount of lower sulfur fuel a ship owner would need to call on the proposed ECA. The hardware costs include additional distillate fuel storage tanks assumed to be constructed of cold rolled steel 1 mm thick and double walled, an LFO fuel separator, an HFO/LFO blending unit, a 3-way valve, an LFO cooler, filters, a viscosity meter, and various pumps and piping. These costs are shown in Table 5.2-11. This cost analysis does not reflect other design options such as partitioning of a residual fuel tank to allow for lower sulfur fuel capacity which would reduce the amount of additional space required, nor does this analysis reflect the possibility that some ships may have already been designed to carry smaller amounts of distillate fuel in separate tanks for purposes other than continuous propulsion.

Table 5.2-11 Fuel Switching Hardware Costs (New Construction)

SPEED	MEDIUM	MEDIUM	MEDIUM	LOW	LOW	LOW
Engine Power (kW)	**4,500**	**9,500**	**18,000**	**8,500**	**15,000**	**48,000**
Cylinders	**9**	**12**	**16**	**6**	**8**	**12**
Liters/cylinder	**35**	**65**	**95**	**380**	**650**	**1400**
Engine Speed (rpm)	**650**	**550**	**500**	**130**	**110**	**100**
Hardware Cost to Supplier						
Component Costs						
Additional Tanks	$3,400	$5,500	$8,300	$4,600	$6,500	$13,700
LFO Separator	$2,800	$3,300	$3,800	$3,800	$4,200	$4,700
HFO/LFO Blending Unit	$4,200	$4,700	$5,600	$4,700	$5,600	$6,600
3-Way Valve	$950	$1,400	$1,900	$1,400	$1,900	$2,800
LFO Cooler	$2,400	$2,800	$3,300	$2,800	$3,800	$4,700
Filters	$950	$950	$950	$950	$950	$950
Viscosity Meter	$1,400	$1,400	$1,400	$1,400	$1,400	$1,400
Piping/Pumps	$2,000	$2,000	$2,000	$2,000	$2,000	$2,000
Total Component Cost	**$18,100**	**$22,100**	**$27,300**	**$21,600**	**$26,400**	**$36,900**
Assembly						
Labor (hours)	240	320	480	320	480	600
Cost ($23.85/hr)	$5,700	$7,600	$11,400	$7,600	$11,400	$14,300
Overhead @ 40%	$2,300	$3,100	$4,600	$3,100	$4,600	$5,700
Total Assembly Cost	**$8,000**	**$10,700**	**$16,000**	**$10,700**	**$16,000**	**$20,000**
Total Variable Cost	$26,100	$32,700	$43,300	$32,300	$42,400	$56,900
Markup @ 29%	$7,600	$9,500	$12,600	$9,400	$12,300	$16,500
Total Hardware RPE	**$33,700**	**$42,200**	**$55,900**	**$41,700**	**$54,700**	**$73,400**
FIXED COSTS						
R&D Costs (0.25 year R&D)	$172,000	$172,000	$172,000	$172,000	$172,000	$172,000
Marine Society Approval	$5,000	$5,000	$5,000	$5,000	$5,000	$5,000
Engines/yr.	40	40	40	40	40	40
Years to recover	5	5	5	5	5	5
Fixed cost/engine	**$880**	**$880**	**$880**	**$880**	**$880**	**$880**

In order to apply the hardware costs associated with the installation of equipment required to use lower sulfur fuel in the proposed ECA, we needed to generate an equation in terms of $/kW that could be applied to other engine sizes. The $/kW value hardware cost values for the six data points corresponding to the six different engine types and sizes used in this analysis were plotted. A curve fit was determined for the slow-speed engine as well as for the medium speed engines, see Figure 5-5.

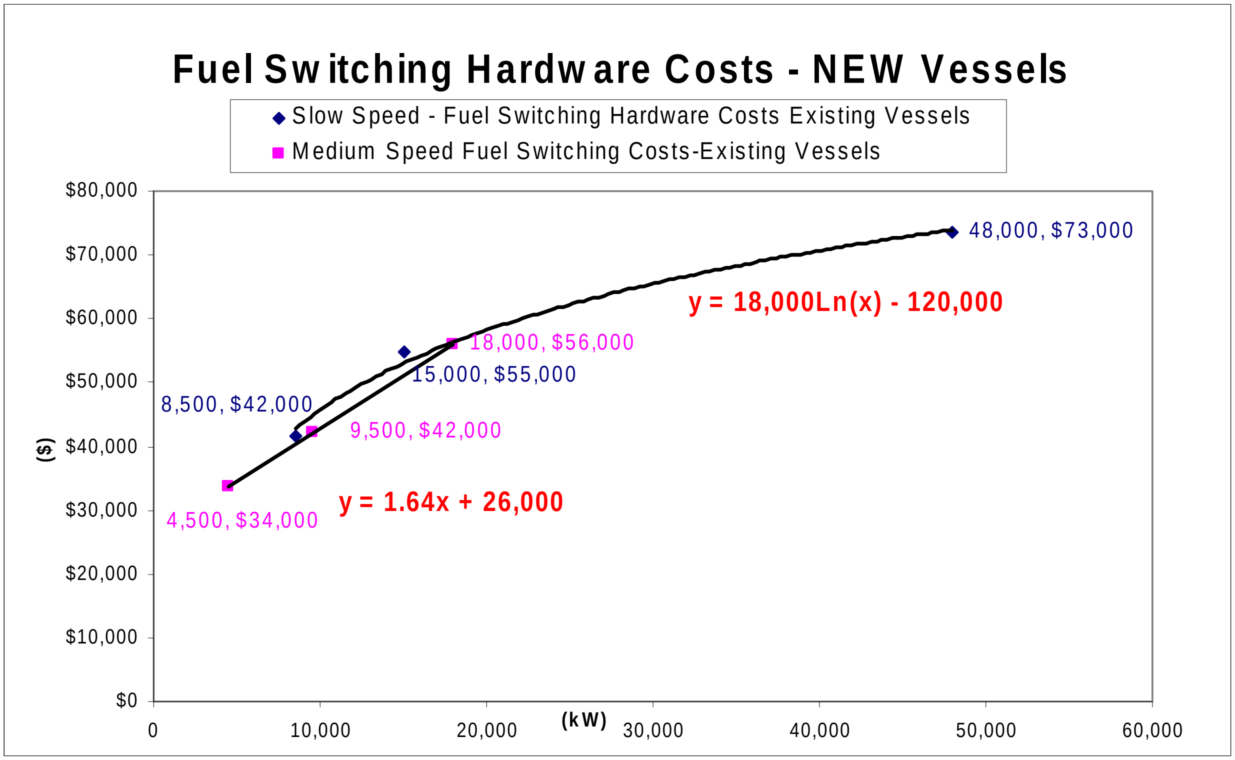

Figure 5-5 $/kW Estimated Hardware Costs Associated with the use of Low Sulfur Fuel

5.2.7 Total Hardware Costs to New Ships in 2020

Total hardware costs associated with the proposed ECA were developed from the number of new ships by ship and engine type estimated to enter the fleet in 2020 as presented earlier in Table 5.2-4. All new vessels were considered to have the average characteristics (including propulsion power) shown in Table 5.2-2. Hardware costs associated with switching to lower sulfur fuel were applied to the percentage[G] of new vessels in 2020 that may require additional tankage, regardless of engine or ship type. The cost estimates developed for the 'typical' engines discussed in Section 5.2.2 were used to develop $/kW equations that could be applied to other engine sizes and types (e.g. SSD and MSD engines). The estimated hardware cost ranges for new vessels, on a per-vessel basis, to meet Tier III NO_X and lower sulfur fuel standards are shown below in Table 5.2-12.

[G] Section 5.1.5 discusses the estimated percentage of the existing fleet that may require modifications to a retrofit, the same percentages were applied to new vessels as it was assumed not all new vessels would require extra hardware to accommodate the use of lower sulfur fuel.

Table 5.2-12 Range of Technology Hardware Costs by Engine Type in $/kW

TECHNOLOGY		ENGINE SPEED	ENGINE SIZE RANGE (KW)	$/KW
SO_X/PM Reductions	Fuel Switching Hardware Costs – *New Vessels*	Medium	4,500 – 18,000	$3.10 - $7.50
		Slow	8,500 – 48,000	$1.50 – $4.90
Tier III NO_X Reductions	SCR Hardware Costs	Medium	4,500 – 18,000	$41.00 - $83.00
		Slow	8,500 – 48,000	$46.00 – $76.00

Table 5.2-13 Total Estimated Variable Hardware Costs per Ship[H]

SHIP TYPE	ENGINE SPEED	AVERAGE PROPULSION POWER (KW)	NEW VESSEL FUEL SWITCHING HARDWARE[a]	AVERAGE PER VESSEL COST OF TIER III[b]
Auto Carrier	MSD	9,600	$42,300	$573,200
Bulk Carrier	MSD	6,400	$36,900	$483,500
Container	MSD	13,900	$49,200	$687,800
General Cargo	MSD	5,200	$34,900	$450,300
Passenger	MSD	23,800	$65,400	$952,500
Reefer	MSD	7,400	$38,500	$511,000
RoRo	MSD	8,600	$40,500	$543,800
Tanker	MSD	6,700	$37,400	$492,800
Misc.	MSD	9,400	$41,900	$566,800
Auto Carrier	SSD	11,300	$48,000	$825,000
Bulk Carrier	SSD	8,400	$42,700	$672,600
Container	SSD	27,500	$63,900	$1,533,100
General Cargo	SSD	7,700	$41,000	$632,900
Passenger	SSD	23,600	$61,200	$1,385,300
Reefer	SSD	10,400	$46,500	$781,000
RoRo	SSD	15,700	$53,900	$1,042,100
Tanker	SSD	9,800	$45,300	$744,200
Misc.	SSD	4,700	$32,000	$453,600

[a] Assumes 32 percent of new vessels would require the fuel switching equipment
[b] The cost estimates presented here represent the average cost per vessel, given that to meet Tier III not all engines are expected to require the same hardware. The costs are determined using the following formula: (5%*($/SHIP_MECH→CR))+(15%*($/SHIP_ELEC→CR))+(T3 ENGINE MODS)+(T3 SCR))

5.2.8 Operational Costs Associated with SCR

In addition to the SCR hardware costs discussed above, ships built as of 2016 would also incur the operating costs associated with SCR's use of urea. The urea operational costs are based on a price of $1.52 per gallon with a density of 1.09 g/cc. The cost per gallon was

[H] Note that not all vessels will need these modifications – it is estimated that only 32% of all vessels will require such additional hardware.

estimated for a 32.5 percent urea solution delivered in bulk to the ship through research completed by ICF combined with historical urea price information.[15,16,17,18] This cost analysis used a urea dosing rate of 7.5 percent that of the brake specific fuel consumption value to estimate how much urea would be used by different engine types and sizes. The total operational costs associated with the proposed ECA are based on the amount of fuel consumed within the proposed ECA in the year 2020. Fuel consumption estimates for 2020 are presented in Chapter 2 of this report including how the amount of fuel used in this area was determined and the fuel costs associated with a U.S. ECA. Based on the U.S. portion of the proposed ECA, the operational costs associated with the use of urea by ships built as of 2016 in 2020 are based on total urea consumption of nearly 100 million gallons are shown in Table 5.2-14 and estimated to be approximately $0.14 billion.

Table 5.2-14 Urea Operational Costs Associated with the use of SCR

SPEED	**MEDIUM**	**MEDIUM**	**MEDIUM**		**LOW**	**LOW**	**LOW**
Engine Power (kW)	**4,500**	**9,500**	**18,000**		**8,500**	**15,000**	**48,000**
Cylinders	**9**	**12**	**16**		**6**	**8**	**12**
Liters/cylinder	**35**	**65**	**95**		**380**	**650**	**1400**
Engine Speed (rpm)	**650**	**550**	**500**		**130**	**110**	**100**
Urea Costs							
BSFC (g/kWh)	210	210	210		195	195	195
Load factor	73%	73%	73%		73%	73%	73%
Aequous Urea Rate	7.5%	7.5%	7.5%		7.5%	7.5%	7.5%
Aqueous Urea (kg/hr)	52	109	207		91	160	512
Aqueous Urea Cost per kg	$0.3684	$0.3684	$0.3684		$0.3684	$0.3684	$0.3684
Aqueous Urea Cost per hour	$19	$40	$76		$33	$59	$188

5.2.9 Existing Vessel Hardware Cost Estimates

This analysis also includes cost estimates for retrofitting existing vessels with additional tankage and related fuel system components, see Table 5.2-15. These hardware costs include additional distillate fuel storage tanks, an LFO fuel separator, an HFO/LFO blending unit, a 3-way valve, an LFO cooler, filters, a viscosity meter, and various pumps and piping as well as additional labor to install the systems on a ship and additional R&D to test systems on existing ships. Similar to the lower sulfur fuel tank analysis discussed above, this existing vessel hardware cost analysis assumes 250 hours of operation, which may be an overestimate of the amount of fuel that is necessary to call on U.S. ports in the ECA. The total estimated hardware costs of retrofitting the portion of the existing fleet estimated to require these modifications is $327 million, these costs would be incurred by 2015.

Table 5.2-15 Fuel Switching Hardware Costs - Existing Vessels

SPEED	MEDIUM	MEDIUM	MEDIUM	LOW	LOW	LOW
Engine Power (kW)	**4,500**	**9,500**	**18,000**	**8,500**	**15,000**	**48,000**
Cylinders	**9**	**12**	**16**	**6**	**8**	**12**
Liters/cylinder	**35**	**65**	**95**	**380**	**650**	**1400**
Engine Speed (rpm)	**650**	**550**	**500**	**130**	**110**	**100**
Hardware Cost to Supplier						
Component Costs						
Additional Tanks	$3,400	$5,500	$8,300	$4,600	$6,500	$13,700
LFO Separator	$2,800	$3,300	$3,800	$3,800	$4,200	$4,700
HFO/LFO Blending Unit	$4,200	$4,700	$5,600	$4,700	$5,600	$6,600
3-Way Valve	$950	$1,400	$1,900	$1,400	$1,900	$2,800
LFO Cooler	$2,400	$2,800	$3,300	$2,800	$3,800	$4,700
Filters	$950	$950	$950	$950	$950	$950
Viscosity Meter	$1,400	$1,400	$1,400	$1,400	$1,400	$1,400
Piping/Pumps	$2,000	$2,000	$2,000	$2,000	$2,000	$2,000
Total Component Cost	**$18,100**	**$22,100**	**$27,300**	**$21,600**	**$26,400**	**$36,900**
Assembly						
Labor (hours)	480	640	960	640	960	1200
Cost ($23.85/hr)	$11,400	$15,300	$22,900	$15,300	$22,900	$28,700
Overhead @ 40%	$4,600	$6,100	$9,200	$6,100	$9,200	$11,400
Total Assembly Cost	**$16,000**	**$21,400**	**$32,100**	**$21,400**	**$32,100**	**$40,100**
Total Variable Cost	$34,100	$43,400	$59,300	$43,00	$58,400	$77,000
Markup @ 29%	$9,900	$12,600	$17,200	$12,500	$17,000	$22,300
Total Hardware RPE	**$44,000**	**$56,000**	**$76,500**	**$55,500**	**$75,400**	**$99,300**
Fixed Costs						
R&D Costs (0.33 year R&D)	$227,000	$227,000	$227,000	$227,000	$227,000	$227,000
Marine Society Approval	$5,000	$5,000	$5,000	$5,000	$5,000	$5,000
Engines/yr.	40	40	40	40	40	40
Years to recover	5	5	5	5	5	5
Fixed cost/engine	**$1,160**	**$1,160**	**$1,160**	**$1,160**	**$1,160**	**$1,160**

5.3 Total Estimated ECA Costs in 2020

The total costs associated with improving ship emissions from current performance to ECA standards in 2020 include both the hardware and operational costs as discussed above. The hardware costs include those of SCR systems and equipment that may be installed on ships built in 2020 to accommodate the use of switching to lower sulfur fuel which together total $1.04 billion in 2020. The operational costs associated with the use of urea are estimated to be $0.14 and the additional fuel costs for the U.S. portion of the proposed ECA will be $1.64 billion in 2020. Therefore, the total costs associated with the U.S. portion of the proposed ECA in 2020 are expected to be $2.78 billion, Table 5.3-1 summarizes these costs.

Table 5.3-1 Total Estimated U.S. ECA Costs in 2020

	TECHNOLOGY	COST IN 2020 (BILLIONS)
Operating Costs (all ships built as of 2016)	Urea Consumption	$0.14
Operating Costs (all ships operating in ECA in 2020)	Fuel Switching	$1.64
Hardware Costs (ships built in 2020)	Fuel Tank Modifications	$0.02
	SCR	$1.02
Total Costs		**$2.78**

5.4 Cost Effectiveness

As discussed in Chapters 3, 4 and 5, the proposed ECA is expected to bring many human health and environmental benefits. Sections 5.1 through 5.3, above, summarize the various costs of the proposed ECA. However, this does not shed light on how cost effective the proposed ECA will be, compared to other control programs, at providing the expected emission reductions.

One tool that can be used to assess the value of the proposed ECA is the measure of cost effectiveness; a ratio of engineering costs incurred per tonne of emissions reduced. The U.S. Government has compared the ECA cost effectiveness to the ratio of costs per tonne of emissions reduced for other control programs. As is shown in this section, the NO_X, SO_X and PM emissions reductions from the proposed ECA compare favorably—in terms of cost effectiveness—to other land-based control programs that have been implemented.

5.4.1 ECA Cost Effectiveness

Chapter 2 of this document summarizes the inventory analyses from which the U.S. projections of pollutant reductions are drawn. The projected U.S. emission reductions due to the proposed ECA are presented above in Table 2-46.

Note that $PM_{2.5}$ is estimated to be 92 percent of the more inclusive PM_{10} emission inventory for marine vessels. In Chapter 2, we generate and present $PM_{2.5}$ inventories since recent research has determined that these are of greater health concern. Traditionally, we have used PM_{10} in our cost effectiveness calculations. Since cost effectiveness is a means of comparing control measures to one another, we use PM_{10} in our cost effectiveness calculations for comparisons to past control measures.

Using the costs associated with NO_X, SO_X and PM control described in sections 5.1 through 5.3 above, and the emission reductions shown in Table 2-46, we calculated the cost per tonne, or cost effectiveness, of the proposed ECA. As described above, the costs of the proposed ECA include costs to refiners to produce additional distillate fuel, as well as costs for engine controls, catalysts and reductants to reduce NO_X emissions and costs for additional tankage for distillate oil. The timing of costs incurred varies, as some costs (i.e. capital expenditures) will be near-term, while others, such as operational costs, are incurred over time in small increments.

The resultant cost per tonne numbers depend on how the costs are allocated to each pollutant. We have allocated costs as closely as possible to the pollutants for which they are incurred. The costs to apply engine controls to meet Tier III NOx standards, including catalysts and reductants, have been allocated to NO_X. In our analyses, we have allocated half of the costs of fuel switching, including production and tankage, to PM and half to SO_X because the costs incurred for control measures to reduce SO_X emissions directly reduce emissions of PM as well.

The resultant estimated cost effectiveness numbers are shown in Table 5.4-1. These include costs and emission reductions that are expected to occur due to compliance with the U.S. portion of the proposed ECA.

Table 5.4-1 Aggregate Long Term ECA Cost per Tonne (2006 U.S. Dollars)

POLLUTANT	30-YR NET PRESENT VALUE DISCOUNTED AT 3%
NO_X	2,600
SO_X	1,200
$PM_{2.5}$	11,000[I]

5.4.2 Land-Based Control Program Cost Effectiveness

The U.S. Government has already imposed restrictions on emissions of NO_X, SO_X, PM and other air pollutants, from a wide range of land-based industrial (stationary) and transportation (mobile) sources as well as consumer and commercial products. We have applied a wide range of programmatic approaches to achieve significant air pollution reductions. Regulatory regimes typically either mandate or incentivize emissions aftertreatment, cleaner fuels or raw materials, improved practices, as well as new processes or technologies.

Significant emission reductions of NO_X and SO_X in the U.S. have been achieved via performance standards for new combustion sources and market-based programs that cap emissions at the regional level. Since 1996, the Acid Rain Program and NO_X Budget Trading Program have been highly successful at drastically reducing both NO_X and SO_X from power plants in the Eastern U.S. Since 2004, NO_X, SO_X and PM emissions from highway and nonroad heavy duty trucks and equipment have been decreasing with performance and emission standards that will be completely phased in by 2010. To allow technology to advance, diesel fuel for use in vehicles in the U.S. and Canada has been reduced to less than 0.0015 percent sulfur (15 parts per million by weight), and diesel fuel for use in off-road equipment, locomotives and domestic marine vessels will be reduced to this level by 2012.

Advanced technology is already required on stationary sources in the U.S., including electricity generation produced by combustion; oil and gas; forest products (including pulp and

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[I] Converting to PM_{10} the cost per tonne would be 10,000. This figure is used in Table 5.4-2 below.

paper and wood products); smelting and refining (including aluminum, alumina, and base metal smelting); iron and steel; iron ore pelletizing; potash; cement; lime; and chemicals production, including fertilizers. On mobile sources, advanced technology to reduce NO_X is phasing in by 2010 for engines on heavy duty trucks and by 2015 for engines on harborcraft.

Programs that are designed to capture the efficiency of designing and building new compliant sources tend to have better cost-effectiveness than programs that principally rely on retrofitting existing sources. Even considering the retrofitting programs, the control measures that have been implemented on land-based sources have been well worthwhile when considering the benefits of the programs. An early example of a highly effective NO_X reduction program is the regional NO_X Budget Program. In 1998, the U.S. Government concluded that NO_X emissions reductions from retrofitting power plants that can be made for less than $3,400 per tonne (in 2006 dollars) are "highly cost effective," considering the emissions reduced by the advanced control technology, not including societal benefits.

The cost of reducing air pollution from these land-based sources has ranged greatly, depending on the pollutant, the type of control program and the nature of the source. A selection of programs and their cost effectiveness is presented in Table 5.4-2. Unless otherwise noted, the programs named in the table address newly built sources only.

Table 5.4-2 Land-Based Source Control Program Cost Per Tonne[a] Comparisons

SOURCE CATEGORY[19]	IMPLEMENTATION DATE	NO_X COST/TONNE	SO_X COST/TONNE	PM_{10} COST/TONNE
Highway Diesel Fuel Program [d] 55 *Fed Reg* 34120, August 21, 1990	1993	-	-	11,000
Stationary Diesel (CI) Engines [c] 71 *Fed Reg* 39154, July 11, 2006	2006	600 - 22,000	-	4,000 - 46,000
Locomotives and Harborcraft (Both New and Retrofits) [d] 73 *Fed Reg* 25097, May 6, 2008	2015	800[b]	-	9,300 (New) 50,000 (Retrofit) [c]
Heavy Duty Nonroad Diesel Engines[d] 69 *Fed Reg* 38957, June 29, 2004	2015	1,200 [b]	900	14,000
Heavy Duty Onroad Diesel Engines [d] 66 *Fed Reg* 5001, January 18, 2001	2010	2,400 [b]	6,400	16,000
International Shipping (ECA) (Both New and Retrofits) [d]	**2016**	**2,600**	**1,200**	**10,000**
Light Duty Gasoline/Diesel Engines [d] 65 *Fed Reg* 6697, February 10, 2000	2009	2,800 [b]	6,600	14,000
Fossil Fuel Fired Power Plants (Retrofits) [c] 58 *Fed Reg* 3590, January 11, 1993; 63 *Fed Reg* 57356, October 27, 1998	2000 to 2010	3,400	300	-
Other Stationary Sources (Both New and Retrofits) [c] 67 *Fed Reg* 80186, December 31, 2002	Ongoing	4,000 - 12,000	300 - 6,000	Variable

Notes:
[a] Units are 2006 U.S. dollars per metric ton. To convert to $/short ton, multiply by 0.907.
[b] Includes NO_X plus non-methane hydrocarbons (NMHC). NMHC are also ozone precursors, thus some rules set combined NO_X+NMHC emissions standards. NMHC are a small fraction of NO_X so aggregate cost/ton comparisons are still reasonable.

[c] Annualized costs of control for individual sources, except SO_X for Power Plants is a typical auction price.
[d] Aggregate program-wide cost/tonne over 30 years, discounted at 3%, except Light Duty and Highway Fuel aggregate costs were discounted at slightly higher rates, yielding slightly lower cost estimates.

Another example of one of the earlier programs is the 1990 regulation promulgated by the U.S. Government to reduce the sulfur content of highway diesel fuel. The cost effectiveness of PM reductions from that program varied depending on how the benefit of reduced wear on the engines was credited. Because the cleaner fuel with 0.05% sulfur (500 ppm) lengthened the useful life of the engines, the program could be characterized as having negative costs (with savings up to $100,000 per tonne) if the maximum engine wear credit was attributed to the program. If no engine wear credit was included, the program was estimated to cost a maximum of $11,000 per tonne of PM reduced.

As shown above, the projected cost per tonne of the proposed ECA falls well within the respective ranges of the other programs. The proposed ECA cost-effectiveness is comparable to the cost per tonne of current programs for new land-based sources, and has favorable cost effectiveness compared to land-based retrofit programs.

[1] Research Triangle Institute, 2008. "Global Trade and Fuels Assessment—Future Trends and Effects of Designating Requiring Clean Fuels in the Marine Sector"; Research Triangle Park, NC; EPA420-R-08-021; November. (Available at http://www.epa.gov/otaq/regs/nonroad/marine/ci/420r08021.pdf)

[2] Research Triangle Institute, 2008. "Global Trade and Fuels Assessment—Future Trends and Effects of Designating Requiring Clean Fuels in the Marine Sector"; Research Triangle Park, NC; EPA420-R-08-021; November. (Available at http://www.epa.gov/otaq/regs/nonroad/marine/ci/420r08021.pdf)

[3] International Maritime Organization, Note by the Secretariat, "Revision of MARPOL Annex VI and NOX Technical Code; Input from the four subgroups and individual experts to the final report of the Informal Cross Government/Industry Scientific Group of Experts," Subcommittee on Bulk Liquids and Gases, 12th Session, Agenda Item 6, BLG 12/INF.10, December 28, 2007.

[4] International Maritime Organization, Note by the Secretariat, "Revision of MARPOL Annex VI and NOX Technical Code; Input from the four subgroups and individual experts to the final report of the Informal Cross Government/Industry Scientific Group of Experts," Subcommittee on Bulk Liquids and Gases, 12th Session, Agenda Item 6, BLG 12/INF.10, December 28, 2007.

[5] EnSys Energy & Systems, Inc. and RTI International 2009. Global Trade and Fuels Assessment—Additional ECA Modeling Scenarios. prepared for the U.S. Environmental Protection Agency.

[6] Energy Information Administration, 2006. "Annual Energy Outlook 2006" (DOE/EIA-0383(2006)); Washington, DC. (Available at: http://www.eia.doe.gov/oiaf/aeo/archive.html)

[7] Energy Information Administration, 2008. "Annual Energy Outlook 2008" (DOE/EIA-0383(2008)); Washington, DC. (Available at: http://www.eia.doe.gov/oiaf/aeo/)

[8] Energy Information Administration, 2008. "International Energy Outlook 2008" (DOE/EIA-0484(2008)); Washington, DC. (Available at: http://www.eia.doe.gov/oiaf/ieo/)

[9] ICF International, "Costs of Emission Reduction Technologies for Category 3 Marine Engines" prepared for the U.S. Environmental Protection Agency, March 2009.

[10] ICF International, "Commercial Marine Port Inventory Development," prepared for the U.S. Environmental Protection Agency, EPA Report Number EPA420-R-07-012c, September 2007.

[11] Lloyd's Register of Ships Online, Lloyd's Register, Fairplay. September, 2008 can be found at www.sea-web.com.

[12] "Matched Typical Ports to Modeled Ports" Table 2-33 of section 2.5.2 of "The Commercial Marine Port Inventory Development, 2002 and 2005 Draft Inventories" report to EPA from ICF International, September 2007.

[13] ICF International, "Commercial Marine Port Inventory Development 2002 and 2005 Draft Inventories" prepared for the U.S. Environmental Protection Agency, September 2007.

[14] Entec UK Limited, Quantification of Emissions from Ships Associated with Ship Movements between Ports in the European Community, July 2002, pps. 86-87.

[15] "Nonroad SCR-Urea Study Final Report" July 29, 2007 TIAX for Engine Manufacturers Association (EMA) can be found at:http://www.enginemanufacturers.org/admin/content/upload/198.pdf

[16] http://www.adblueonline.co.uk/air_1/bulk_delivery

[17] http://www.factsaboutscr.com/documents/IntegerResearch-Ureapricesbackto2005levels.pdf

[18] http://www.fertilizerworks.com/fertreport/index.html

[19] Regulation of Fuels and Fuel Additives: Fuel Quality Regulations for Highway Diesel Fuel Sold in 1993 and Later Calendar Years, 55 *Fed Reg* 34120, August 21, 1990.
Standards of Performance for Stationary Compression Ignition Internal Combustion Engines, 71 *Fed Reg* 39154, July 11, 2006.
Control of Emissions of Air Pollution from Locomotives and Marine Compression-Ignition Engines Less Than 30 Liters per Cylinder, 73 *Fed Reg* 25097, May 6, 2008.
Control of Emissions of Air Pollution From Nonroad Diesel Engines and Fuel 69 *Fed Reg* 38957, June 29, 2004.
Control of Air Pollution from New Motor Vehicles: Heavy-Duty Engine and Vehicle Standards and Highway Diesel Fuel Sulfur Control Requirements 66 *Fed Reg* 5001, January 18, 2001.
Control of Air Pollution From New Motor Vehicles: Tier 2 Motor Vehicle Emissions Standards and Gasoline Sulfur Control Requirements 65 *Fed Reg* 6697, February 10, 2000.
Acid Rain Program; General Provisions and Permits, Allowance System, Continuous Emissions Monitoring, Excess Emissions and Administrative Appeals, 58 *Fed Reg* 3590, January 11, 1993; Finding of Significant Contribution and Rulemaking for Certain States in the Ozone Transport Assessment Group Region for Purposes of Reducing Regional Transport of Ozone, 63 *Fed Reg* 57356, October 27, 1998.
Prevention of Significant Deterioration (PSD) and Nonattainment New Source Review (NSR): Baseline Emissions Determination, Actual-to-Future-Actual Methodology, Plantwide Applicability Limitations, Clean Units, Pollution Control Projects, 67 *Fed Reg* 80186, December 31, 2002

6 Economic Impacts

Chapter 5 provides the engineering costs associated with complying with the Tier III NO_X limits and the ECA fuel sulfur limits for all ships operating in the U.S. portion of the proposed ECA in 2020. In this chapter, we examine the economic impacts of these costs on shipping engaged in international trade. We look at two aspects of the economic impacts: estimated social costs and how they are shared across stakeholders, and estimated market impacts in terms of changes in prices and quantities produced for directly affected markets. All costs are presented in terms of 2006 U.S. dollars.

The total estimated social costs associated with the U.S. portion of the proposed ECA in 2020 are equivalent to the estimated compliance costs of the program, at approximately $2.78 billion. These costs are expected to accrue initially to the owners and operators of affected vessels. These owners and operators are expected to pass their increased costs on to the entities that purchase their transportation services in the form of higher freight rates. Ultimately, these costs will be borne by the final consumers of goods transported by ocean-going vessels in the form of higher prices for those goods.

The compliance costs associated with the U.S. portion of the proposed ECA are described earlier in this chapter. We estimate that these costs added to the total cost of shipping goods to or from a U.S. origin or destination will result in only a modest increase in the costs of goods transported by ship. We estimate that the cost to comply with the ECA requirements would increase the price of a new vessel by 2 percent or less. With regard to operating costs, analysis of a ship in liner service between Singapore, Seattle, and Los Angeles/Long Beach, which includes about 1,700 nm of operation in the proposed ECA, suggests that improving from current performance to ECA standards would increase the operating costs by about 3 percent. For a container ship, this represents a price increase of about $18 per container, assuming the total increase in operating costs is passed on to the purchaser of marine transportation services. This would be about a 3 percent price increase. The per passenger price of a seven-day Alaska cruise operating entirely within the ECA is expected to increase about $7 per day. For ships that spend less time in the ECA, the expected increase in total operating costs would be smaller.

It should be noted that this economic analysis holds all other aspects of the market constant except for the designation of the proposed ECA. It does not attempt to predict the equilibrium market conditions for 2020, particularly with respect to how excess capacity in today's market due to the current economic downturn will be absorbed. This approach is appropriate because the goal of an economic impact analysis is to explore the impacts of a specific program; allowing changes in other market conditions would confuse the impacts due to the proposed regulatory program.

The remainder of this chapter provides detailed information on the methodology we used to estimate these economic impacts and the results of our analysis.

6.1 The Purpose of an Economic Impact Analysis

An Economic Impact Analysis (EIA) is prepared to provide information about the potential economic consequences of a regulatory action. Such an analysis consists of estimating the social costs of a regulatory program and the distribution of these costs across stakeholders.

In an economic impact analysis, social costs are the value of the goods and services lost by society resulting from a) the use of resources to comply with and implement a regulation and b) reductions in output. There are two parts to the analysis. In the economic welfare analysis, we look at the total social costs associated with the program and their distribution across key stakeholders. In the market analysis, we estimate how prices and quantities of goods and directly affected by the emission control program can be expected to change once the program goes into effect.

6.2 Economic Impact Analysis Methodology

Economic impact analysis is rooted in basic microeconomic theory. We use the laws of supply and demand to simulate how markets can be expected to respond to increases in production costs that occur as a result of the new emission control program. Using that information, we construct the social costs of the program and identify how those costs will be shared across the markets and, thus, across stakeholders. The relevant concepts are summarized below and are presented in greater detail in Appendix 6A to this chapter.

Before the implementation of a control program, a market is assumed to be in equilibrium, with producers producing the amount of a good that consumers desire to purchase at the market price. The implementation of a control program results in an increase in production costs by the amount of the compliance costs. This generates a "shock" to the initial equilibrium market conditions (a change in supply). Producers of affected products will try to pass some or all of the increased production costs on to the consumers of these goods through price increases, without changing the quantity produced. In response to the price increases, consumers will decrease the quantity they buy of the affected good (a change in the quantity demanded). This creates surplus production at the new price. Producers will react to the decrease in quantity demanded by reducing the quantity they produce, and they will be willing to sell the remaining production at a lower price that does not cover the full amount of the compliance costs. Consumers will then react to this new price. These interactions continue until the surplus is removed and a new market equilibrium price and quantity combination is achieved.

The amount of the compliance costs that will be borne by stakeholders is ultimately limited by the price sensitivity of consumers and producers in the relevant market, represented by the price elasticities of demand and supply for each market. An "inelastic" price elasticity (less than one) means that supply or demand is not very responsive to price changes (a one percent change in price leads to less than one percent change in quantity). An "elastic" price elasticity (more than one) means that supply or demand is sensitive to price changes (a one percent change in price leads to more than one percent change in quantity). A price elasticity of one is unit elastic, meaning there is a one-to-one correspondence between a percent change in price and percent change in quantity.

On the production side, price elasticity of supply depends on the time available to adjust production in response to a change in price, how easy it is to store goods, and the cost of increasing (or decreasing) output. In this analysis we assume the supply for engines, vessels, and marine transportation services is elastic: an increase in the market price of an engine, vessel or freight rates will lead producers to want to produce more, while a decrease will lead them to produce less (this is the classic upward-sloping supply curve). It would be difficult to estimate the slope of the supply curve for each of these markets given the global nature of the sector. However, it is reasonable to assume that the supply elasticity for the ocean marine transportation services market is likely to be greater than one. This is because output can more easily be adjusted due to a change in price. For the same reason, the supply elasticity for the new Category 3 engine market is also likely to be greater than one, especially since these engines are often used in other land-based industries, especially in power plants. The supply elasticity for the vessel construction market, on the other hand, may be less than or equal to one, depending on the vessel type, since it may be harder to adjust production and/or store output if the price drops, or rapidly increase production if the price increases. Because of the nature of this industry, it would not be possible to easily switch production to other goods, or to stop or start production of new vessels.

On the consumption side, we assume that the demand for engines is a function of the demand for vessels, which is a function of the demand for international shipping (demand for engines and vessels is derived from the demand for marine transportation services). This makes intuitive sense: Category 3 engine and ocean-going vessel manufacturers would not be expected to build an engine or vessel unless there is a purchaser, and purchasers will want a new vessel/engine only if there is a need for one to supply marine transportation services. Deriving the price elasticity of demand for the vessel and engine markets from the international shipping market is an important feature of this analysis because it provides a link between the product markets.

In this analysis, the price elasticity of demand is nearly perfectly inelastic. This stems from the fact that, that, for most goods, there are no reasonable alternative shipping modes. In most cases, transportation by rail or truck is not feasible, and transportation by aircraft is too expensive. Approximately 90 percent of world trade by tonnage is moved by ship, and ships provide the most efficient method to transport these goods on a tonne-mile basis.[1] Stopford notes that "shippers need the cargo and, until they have time to make alternative arrangements, must ship it regardless of cost … The fact that freight generally accounts for only a small portion of material costs reinforces this argument."[2] A nearly perfectly inelastic price elasticity of demand for marine transportation services means that virtually all of the compliance costs can be expected to be passed on to the consumers of marine transportation services, with no change in output for engine producers, ship builders, or owners and operators of ships engaged in international trade.

The economic impacts described below rely on the estimated engineering compliance costs presented in Chapter 5. These include the cost of hardware for new vessels to comply with the Tier III engine standards, and the cost of fuel switching equipment for certain new and existing vessels. Also included are expected increases in operating costs for vessels operating in the ECA. These increased operating costs include changes in fuel consumption rates, increases in fuel costs, and the use of urea for engines equipped with SCR, as well as a small increase in operating costs for operation outside the ECA due to the fuel price impacts of the program.

6.3 Expected Economic Impacts of the Proposed ECA

6.3.1 Engine and Vessel Market Impacts

The assumption of nearly perfectly inelastic demand for marine transportation services means that the amount of these services purchased is not expected to change as a result of costs of complying with the ECA requirements in the U.S. portion of the proposed ECA. As a result, the demand for vessels and engines would also not change compared to the no-control scenario, and the quantities produced would stay the same in 2020.

Also due to the assumption of nearly perfectly inelastic demand for marine transportation services, the price impacts would be equivalent to the engineering compliance costs for the new engine and vessel markets. Estimated price impacts for a sample of engine and vessel combinations are set out in Table 6.3-1, for medium speed engines, and Table 6.3-2, for slow speed engines.

Table 6.3-1 Summary of Estimated Market Impacts – New Medium Speed Engines and Vessels (2020; $2006)

SHIP TYPE	AVERAGE PROPULSION POWER	NEW VESSEL ENGINE PRICE IMPACT (NEW TIER III ENGINE PRICE IMPACT)[A]	NEW VESSEL FUEL SWITCHING EQUIPMENT PRICE IMPACT[B]	NEW VESSEL TOTAL PRICE IMPACT
Auto Carrier	9,600	$573,200	$42,300	$615,500
Bulk Carrier	6,400	$483,500	$36,900	$520,400
Container	13,900	$687,800	$49,200	$736,000
General Cargo	5,200	$450,300	$34,900	$475,200
Passenger	23,800	$952,500	$65,400	$1,107,900
Reefer	7,400	$511,000	$38,500	$549,500
RoRo	8,600	$543,800	$40,500	$584,300
Tanker	6,700	$492,800	$37,400	$530,200
Misc.	9,400	$566,800	$41,900	$608,700

[a] Medium speed engine price impacts are estimated from the cost information presented in Chapter 5 using the following formula: (10%*($/SHIP_MECH→CR))+(30%*($/SHIP_ELEC→CR))+(T3 ENGINE MODS)+(T3SCR))
[b] Assumes 32 percent of new vessels would require the fuel switching equipment.

These price impacts reflect the impacts of the costs that will be incurred when the most stringent ECA standards are in place in 2020. These estimated price impacts are small when compared to the price of a new vessel.

Table 6.3-2 Summary of Estimated Market Impacts – Slow Speed Engines and Vessels (2020; $2006)

SHIP TYPE	AVERAGE PROPULSION POWER	NEW VESSEL ENGINE PRICE IMPACT (NEW ENGINE PRICE IMPACT)[A]	NEW VESSEL FUEL SWITCHING EQUIPMENT PRICE IMPACT[B]	NEW VESSEL TOTAL PRICE IMPACT
Auto Carrier	11,300	$825,000	$48,000	$873,000
Bulk Carrier	8,400	$672,600	$42,700	$715,300
Container	27,500	$1,533,100	$63,900	$1,597,000
General Cargo	7,700	$632,900	$41,000	$673,900
Passenger	23,600	$1,385,300	$61,200	$1,446,500
Reefer	10,400	$781,000	$46,500	$827,500
RoRo	15,700	$1,042,100	$53,900	$1,096,000
Tanker	9,800	$744,200	$45,300	$789,500
Misc.	4,700	$453,600	$32,000	$485,600

[a] Slow speed engine price impacts are estimated from the cost information presented in Chapter 5 using the following formula: (5%*($/SHIP_MECH→CR))+(15%*($/SHIP_ELEC→CR))+(T3 ENGINE MODS)+(T3 SCR))

[b] Assumes 32 percent of new vessels would require the fuel switching equipment

A selection of new vessel prices is provided in Table 6.3-3, and range from about $40 million to $480 million. The program price increases range from about $600,000 to $1.5 million. A price increase of $600,000 to comply with the ECA requirements would be an increase of approximately 2 percent for a $40 million vessel. The largest vessel price increase noted above, for a passenger vessels, is about $1.5 million; this is a price increase of less than 1 percent for a $478 million passenger vessel. Independent of the nearly perfect inelasticity of demand, price increases of this magnitude would be expected to have little, if any, effect on the quantity sales of new vessels, all other economic conditions held constant.

Table 6.3-3 Newbuild Vessel Price by Ship Type and Size, Selected Vessels (Millions, $2008)

VESSEL TYPE	VESSEL SIZE CATEGORY	SIZE RANGE (MEAN) (DWT)	NEWBUILD
Bulk Carrier	Handy	10,095 – 39,990 (27,593)	$56.00
	Handymax	40,009 – 54,881 (47,616)	$79.00
	Panamax	55,000 – 78,932 (69,691)	$97.00
	Capesize	80,000 – 364,767 (157,804)	$175.00
Container	Feeder	1,000-13,966 (9,053)	$38.00
	Intermediate	14,003-36,937 (24,775)	$70.00
	Panamax	37,042-54,700 (45,104)	$130.00
	Post Panamax	55,238-84,900 (67,216)	$165.00
Gas carrier	Midsize	1,001-34,800 (7,048)	$79.70
	LGC	35,760-59,421 (50,796)	$37.50
	VLGC	62,510-122,079 (77,898)	$207.70
General cargo	Coastal Small	1,000-9,999 (3,789)	$33.00
	Coastal Large	10,000-24,912 (15,673)	$43.00
	Handy	25,082-37,865 (29,869)	$52.00
	Panamax	41,600-49,370 (44,511)	$58.00

VESSEL TYPE	VESSEL SIZE CATEGORY	SIZE RANGE (MEAN) (DWT)	NEWBUILD
Passenger	All	1,000–19,189 (6,010)	$478.40
Reefer	All	1,000–19,126 (6,561)	$17.30
Ro-Ro	All	1,000–19,126 (7,819)	$41.20
Tanker	Coastal	1,000-23,853 (7,118)	$20.80
	Handymax	25,000-39,999 (34,422)	$59.00
	Panamax	40,000-75,992 (52,300)	$63.00
	AFRAmax	76,000-117,153 (103,112)	$77.00
	Suezmax	121,109-167,294 (153,445)	$95.00
	VLCC	180,377-319,994 (294,475)	$154.00
Sources: Lloyd's Shipping Economist (2008), Informa (2008), Lloyd's Sea-Web (2008)			

6.3.2 Fuel Market Impacts

The market impacts for the fuel markets were estimated through the modeling performed to estimate the fuel compliance costs for the coordinated strategy. In the WORLD model, the total quantity of fuel used is held constant, which is consistent with the assumption that the demand for international shipping transportation would not be expected to change due to the lack of transportation alternatives.

The expected price impacts of the coordinated program are set out in Table 6.3-4. Note that on a mass basis, less distillate than residual fuel is needed to go the same distance (5 percent less). The prices in Table 6.3-4 are adjusted for this impact.

Table 6.3-4 shows that the coordinated strategy is expected to result in a small increase in the price of marine distillate fuel, about 1.3 percent. The price of residual fuel is expected to decrease slightly, by less than one percent, due to a reduction in demand for that fuel.

Table 6.3-4 Summary of Estimated Market Impacts - Fuel Markets

FUEL	UNITS	BASELINE PRICE	CONTROL PRICE	ADJUSTED FOR ENERGY DENSITY	% CHANGE
Distillate	$/tonne	$462	$468	N/A	+1.3%
Residual	$/tonne	$322	$321	N/A	-0.3%
Fuel Switching	$/tonne	$322	$468	$444	+38.9%

Because of the need to shift from residual fuel to distillate fuel in the ECA, ship owners are expected to see an increase in their total cost of fuel. This increase is because distillate fuel is more expensive than residual fuel. Factoring in the higher energy content of distillate fuel, relative to residual fuel, the fuel cost increase would be about 39 percent.

6.3.3 Marine Transportation Market Impacts

We used the above information to estimate the impacts on the prices of marine transportation services. This analysis, presented in Appendix 6B to this chapter, is limited to the impacts of increases in operating costs due to the fuel and emission requirements of the coordinated strategy. Operating costs would increase due to the increase in the price of fuel, the need to switch to fuel with a sulfur content not to exceed 1,000 ppm while operating in the ECA, and due to the need to dose the aftertreatment system with urea to meet the Tier III standards.

Estimates of the impacts of these increased operating costs were performed using a representative fleet, fuel cost, actual operational parameters, and sea-route data for three types of ocean going vessels: container, bulk carrier, and cruise liner. The representative fleet values used were obtained from the Lloyd's of London Sea-Web Database, and were based on actual vessel size (Dead Weight Tonnes (DWT)) and engine power (kilowatt – hour (kW-hr)) of each vessel type. Additionally, to develop a representative sea-route for our price estimations, we created two theoretical trips, a 'circle route' occurring in the Pacific Ocean and an Alaskan cruise. The total nautical mileage (nm) for the 'circle route' was determined to be 15,876 nm, with approximately 1,700 nm occurring within the proposed U.S. ECA boundary, while the Alaskan voyage travelled up the Canadian / Alaskan coastline for seven days, stopping at five destinations, and operating completely in the proposed ECA for a total of 2,000 nm. We also estimated the impacts for a trip to the port at Montreal (1,000 nm).

To conduct our price increase estimations, we calculated the average fuel operational costs of the theoretical 'circle route' for the container and bulk carrier, and the Alaskan voyage for the cruise liner as they would function today, completely on residual fuel. We then calculated the operational fuel costs for the vessels if they were to travel the route with the U.S. ECA in place. This ECA calculation was conducted assuming that the vessel would continue to operate on residual fuel when outside of the ECA, and that approximately 33 percent of these vessels would also use an exhaust aftertreatment technology that would require urea usage.

The overall price differences for each of these hypothetical trips were obtained by subtracting the residual fuel operational costs from the calculated ECA operational fuel / urea costs. Table 6.3-5 summarizes these price increases as they relate to goods shipped and per-passenger impacts. Additionally, the table lists the vessel and engine parameters that were used in the calculations.

Table 6.3-5 Summary of Impacts of Operational Fuel / Urea Cost Increases

VESSEL TYPE	VESSEL AND ENGINE PARAMETERS	OPERATIONAL PRICE INCREASES
Container North Pacific Circle Route	36,540 kW 50,814 DWT	$17.53/TEU
Bulk Carrier North Pacific Circle Route	3,825 kW 16,600 DWT	$0.56 / tonne
Cruise Liner (Alaska)	31,500 kW 226,000 DWT 1,886 passengers	$6.60 / per passenger per day

This information suggests that the increase in marine transportation service prices would be small, both absolutely and when compared to the price charged by the ship owner per unit transported. For example, Stopford notes that the price of transporting a 20 foot container between the UK and Canada is estimated to be about $1,500; of that, $700 is the cost of the ocean freight; the rest is for port, terminal, and other charges.[3] An increase of about $18 represents an increase of less than 3 percent of ocean freight cost, and about one percent of transportation cost. Similarly, the price of a 7-day Alaska cruise varies from $100 to $400 per night or more. In that case, this price increase would range from 1.5 percent to about 6 percent.

Our analysis also suggests that increases in operational costs of the magnitude expected to occur for vessels operating in the ECA are within the range of historic price variations for bunker fuel. This is illustrated in Figure 6.3-1. This figure is based on variation in fuel price among the ports of Singapore, Houston, Rotterdam, and Fujairah.

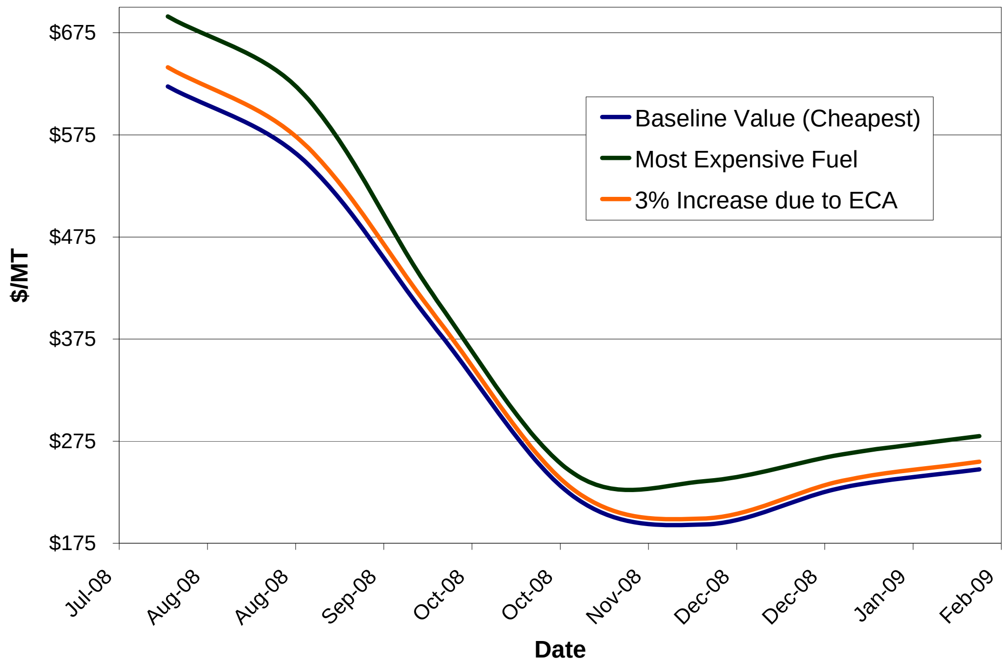

Figure 6.3-1 Range of Bunker Fuel Prices

This graph illustrates the price differential between these ports, comparing the estimated 3% ECA increase to the cheapest fuel for each month. We then plotted these calculated ECA increases (the 3% increases), the cheapest fuel (as a baseline) and the most expensive fuel for the same six month period. As can be observed from the previous calculations and the trends in Figure 1, there are both spatial and temporal price fluctuations in fuel prices. During this period (granted, a period of above-average fluctuations), the price of fuel varied both spatially and temporally. The variation over time is higher than the variation over ports; however, by either form of variation, the 3% increase in bunker fuel price due to the ECA is smaller than the normal price variation of the fuel.

6.3.4 Social Costs of the Proposed ECA and Distribution Across Stakeholders

The total social costs associated with complying with the Tier III NO_X limits and the ECA fuel sulfur limits for all ships operating in the U.S. portion of the proposed ECA are estimated to be the same as the total engineering costs presented in Chapter 5, or about $2.78 billion in 2020. For the reasons described above and explained more fully in the Appendix to this chapter, these costs are expected to be borne fully by consumers of international shipping services.

These social costs are small when compared to the total value of U.S. waterborne foreign trade. In 2007, waterborne trade for government and non-government shipments by vessel into and out of U.S. foreign trade zones, the 50 states, the District of Columbia, and Puerto Rico was about $1.4 trillion. Of that, about $1 trillion was for imports.[4]

Appendices

Appendix 6A

The methodology used in this Economic Impact Analysis (EIA) is rooted in applied microeconomic theory and was developed following U.S. EPA's recommended procedures.[5] This appendix describes the economic theory underlying the analysis and how it was applied to the problem of estimating the economic impacts of the proposed ECA on shipping engaged in international trade.

The Economic Theory Used to Estimate Economic Impacts

The approach used to estimate the economic impacts of the proposed ECA relies on the basic relationships between production and consumption in competitive markets.

Multi-Market, Partial-Equilibrium Approach

The approach is *behavioral* in that it builds on the engineering cost analysis by incorporating economic theory related to producer and consumer behavior to estimate changes in market conditions. As Bingham and Fox[6] note, this framework provides "a richer story" of the expected distribution of economic welfare changes across producers and consumers. In behavioral models, manufacturers of goods affected by a regulation are economic agents who can make adjustments, such as changing production rates or altering input mixes, which will generally affect the market environment in which they operate. As producers change their production levels in response to a new regulation, consumers of the affected goods are typically faced with changes in prices that cause them to alter the quantity that they are willing to purchase. These changes in price and output resulting from the market adjustments are used to estimate the distribution of social costs between consumers and producers.

This is also a *multi-market, partial equilibrium* approach. It is a multi-market approach in that more than one market is examined: the markets for marine engines, vessels, and international shipping transportation services. It is a partial-equilibrium approach in that rather than explicitly modeling all of the interactions in the global economy that are affected by international shipping, the individual markets that are directly affected by the ECA requirements are modeled in isolation. This technique has been referred to in the literature as "partial equilibrium analysis of multiple markets."[7]

This EIA does not examine the economic impact of the proposed ECA on finished goods that use ocean transportation services as inputs. This is because international shipping transportation services are only a small part of the total inputs of the final goods and services produced using the materials shipped. A change in the price of marine transportation services on the order anticipated by this program would not be expected to significantly affect the markets for the finished goods. So, for example, while we look at the impacts of the program on ocean transportation costs, we do not look at the impacts of the controls on gasoline produced using crude oil transported by ship, or on manufactured products that use petroleum products as inputs.

It should also be noted that this EIA estimates the aggregate economic impacts of the control program at the market level. This is not intended to be a firm-level analysis; therefore compliance costs facing any particular ship operator may be different from the market average, and the impacts of the program on particular firms can vary significantly. The difference can be important, particularly where the rule affects different firms' costs over different activity rates.

Competitive Markets

The methodology used in this EIA relies on an assumption of perfect competition. This means that consumers and firms are price takers and do not have the ability to influence market prices. Perfect competition is widely accepted for this type of analysis and only in rare cases are other approaches used.[8] Stopford's description of the shipping market and how prices are set in this market supports this assumption.[9]

In a perfectly competitive market at equilibrium with no externalities, the market price equals the value society (consumers) places on the marginal product, as well as the marginal cost to society (producers). Producers are price takers, in that they respond to the value that consumers put on the product. It should be noted that the perfect competition assumption is not primarily about the number of firms in a market. It is about how the market operates: whether or not individual firms have sufficient market power to influence the market price. Indicators that allow us to assume perfect competition include absence of barriers to entry, absence of strategic behavior among firms in the market, and product differentiation.[J,10] Finally, according to contestable market theory, oligopolies and even monopolies will behave very much like firms in a competitive market if it is possible to enter particular markets costlessly (i.e., there are no sunk costs associated with market entry or exit). This would be the case, for example, when products are substantially similar (e.g., a recreational vessel and a commercial vessel).

Intermediate-Run Impacts

This EIA explores economic impacts on affected markets in the intermediate run. In the intermediate run, some factors of production are fixed and some are variable. A short-run analysis, in contrast, imposes all compliance costs on producers, while a long-run analysis imposes all costs on consumers. The use of the intermediate run means that some factors of production are fixed and some are variable, and illustrates how costs will be shared between producers and consumers as the markets adjust to the new compliance program. The use of the intermediate time frame is consistent with economic practices for this type of analysis.

Short-Run Analysis

In the very short run, all factors of production are assumed to be fixed, leaving producers with no means to respond to the increased costs associated with the regulation (e.g., they cannot adjust labor or capital inputs). Within a very short time horizon, regulated producers are constrained in their ability to adjust inputs or outputs due to contractual, institutional, or other

[J] The number of firms in a market is not a necessary condition for a perfectly competitive market. See Robert H. Frank, Microeconomics and Behavior, 1991, McGraw-Hill, Inc., p 333.

factors and can be represented by a vertical supply curve, as shown in Figure 6A-1. Under this time horizon, the impacts of the regulation fall entirely on the regulated entity. Producers incur the entire regulatory burden as a one-to-one reduction in their profit. This is referred to as the "full-cost absorption" scenario and is equivalent to the engineering cost estimates. Although there is no hard and fast rule for determining what length of time constitutes the very short run, it is inappropriate to use this time horizon for this type of analysis because it assumes economic entities have no flexibility to adjust factors of production. Note that the BAF is a way to avoid this scenario. Additionally, the fact that liner price schedules are renegotiated at least annually, and that individual service contracts may be negotiated more frequently, suggests that a very short-run analysis would not be suitable.

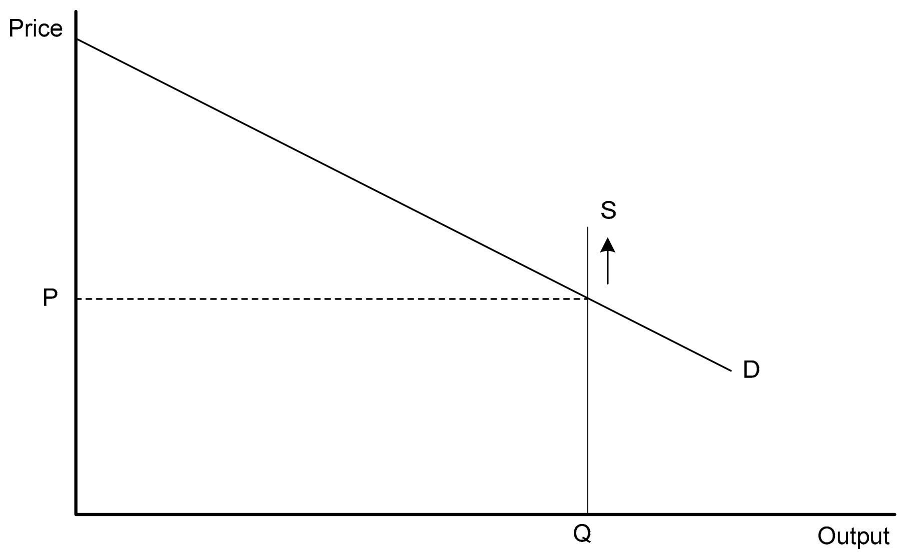

Figure 6A-1 Short-Run: All Costs Borne by Producers

Long-Run Analysis

In the long run, all factors of production are variable, and producers can be expected to adjust production plans in response to cost changes imposed by a regulation (e.g., using a different labor/capital mix). Figure 6A-2 illustrates a typical, if somewhat simplified, long-run industry supply function. The supply function is horizontal, indicating that the marginal and average costs of production are constant with respect to output. This horizontal slope reflects the fact that, under long-run constant returns to scale, technology and input prices ultimately determine the market price, not the level of output in the market.

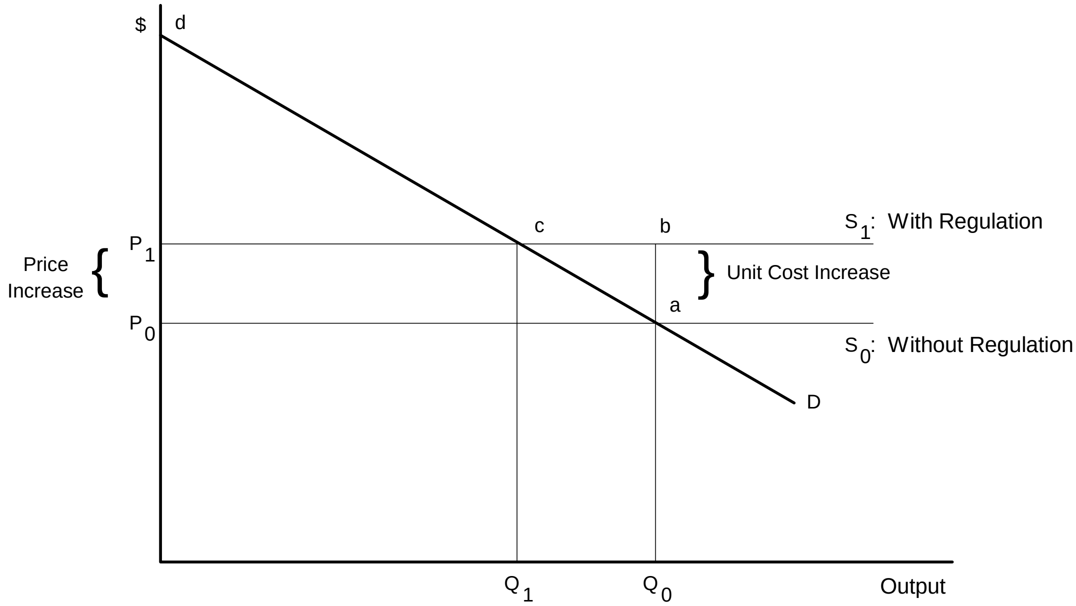

Figure 6A-2 Long-Run: Full Cost Pass-Through

Market demand is represented by the standard downward-sloping curve. The market is assumed here to be perfectly competitive; equilibrium is determined by the intersection of the supply and demand curves. In this case, the upward shift in the market supply curve represents the regulation's effect on production costs and is illustrated in Figure 6A-2. The shift causes the market price to increase by the full amount of the per-unit control cost (i.e., from P_0 to P_1). With the quantity demanded sensitive to price, the increase in market price leads to a reduction in output in the new with-regulation equilibrium (i.e., Q_0 to Q_1). As a result, consumers incur the entire regulatory burden as represented by the loss in consumer surplus (i.e., the area P_0ac P_1). In the nomenclature of EIAs, this long-run scenario is typically referred to as "full-cost pass-through."

Taken together, impacts modeled under the long-run/full-cost-pass-through scenario reveal an important point: under fairly general economic conditions, a regulation's impact on producers is transitory. Ultimately, the costs are passed on to consumers in the form of higher prices. However, this does not mean that the impacts of a regulation will have no impact on producers of goods and services affected by a regulation. For example, the long run may cover the time taken to retire today's entire capital equipment, which could take decades. Therefore, transitory impacts could be protracted and could dominate long-run impacts in terms of present value. In addition, to evaluate impacts on current producers, the long-run approach is not appropriate. Consequently a time horizon that falls between the very short-run/full-cost-absorption case and the long-run/full-cost-pass-through case is most appropriate for this EIA.

Intermediate Run Analysis

The intermediate run time frame allows examination of impacts of a regulatory program during the transition between the very short run and the long run. In the intermediate run, there is some resource immobility which may cause producers to suffer producer surplus losses. Specifically, producers may be able to adjust some, but not all, factors of production, and they therefore will bear some portion of the costs of the regulatory program. The existence of fixed production factors generally leads to diminishing returns to those fixed factors. This typically manifests itself in the form of a marginal cost (supply) function that rises with the output rate, as shown in Figure 6A-3.

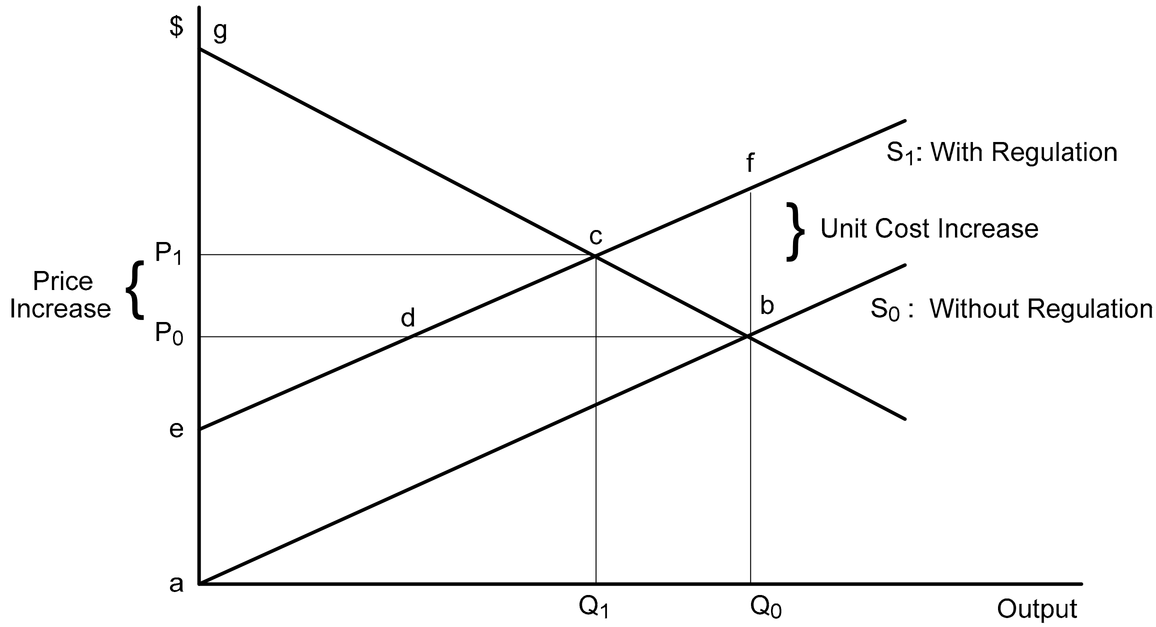

Figure 6A-3 Intermediate-Run: Partial-Cost Pass-Through

Again, the regulation causes an upward shift in the supply function. The lack of resource mobility may cause producers to suffer profit (producer surplus) losses in the face of regulation; however, producers are able to pass through some of the associated costs to consumers, to the extent the market will allow. As shown, in this case, the market-clearing process generates an increase in price (from P_0 to P_1) that is less than the per-unit increase in costs, so that the regulatory burden is shared by producers (net reduction in profits) and consumers (rise in price). In other words, there is a loss of both producer and consumer surplus.

Economic Impacts of a Control Program – Single Market

A graphical representation of a general economic competitive model of price formation, as shown in Figure 6A-4(a), posits that market prices and quantities are determined by the intersection of the market supply and market demand curves. Under the baseline scenario, a market price and quantity (p,Q) are determined by the intersection of the downward-sloping market demand curve (D^M) and the upward-sloping market supply curve (S^M). The market supply curve reflects the sum of the domestic (S_d) and import (S_f) supply curves.

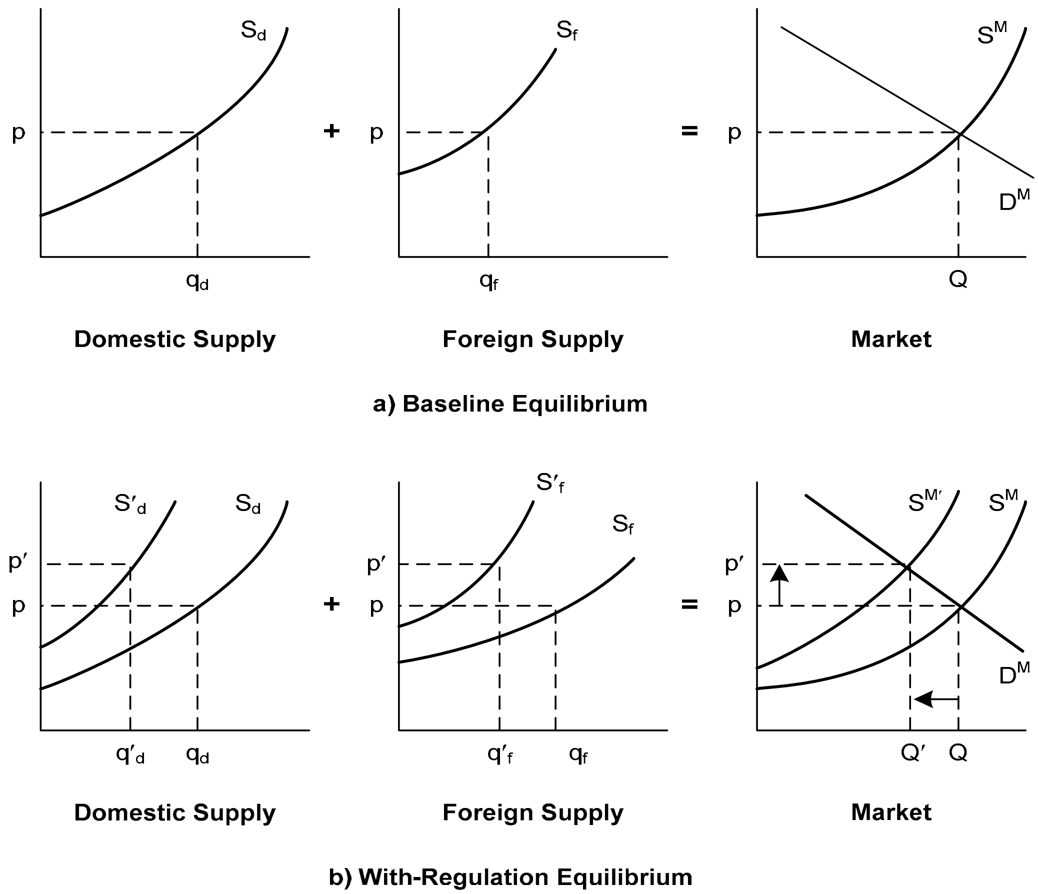

Figure 6A-4 Market Equilibrium Without and With Regulation

With the regulation, the costs of production increase for suppliers. The imposition of these regulatory control costs is represented as an upward shift in the supply curve for domestic and import supply by the estimated compliance costs. As a result of the upward shift in the supply curve, the market supply curve will also shift upward as shown in Figure 6A-4(b) to reflect the increased costs of production.

At baseline without the new standards, the industry produces total output, Q, at price, p, with domestic producers supplying the amount q_d and imports accounting for Q minus q_d, or q_f. With the regulation, the market price increases from p to p′, and market output (as determined from the market demand curve) decreases from Q to Q′. This reduction in market output is the net result of reductions in domestic and import supply.

As indicated in Figure 6A-4, when the new standards are applied the supply curve will shift upward by the amount of the estimated compliance costs. The demand curve, however, does not shift in this analysis. This is explained by the dynamics underlying the demand curve. The demand curve represents the relationship between prices and quantity demanded. Changes in prices lead to changes in the quantity demanded and are illustrated by *movements along* a constant demand curve. In contrast, changes in consumer tastes, income, prices of related goods, or population would lead to change in demand and are illustrated as *shifts* in the position of the demand curve.[K,11] For example, an increase in the number of consumers in a market would cause the demand curve to shift outward because there are more individuals willing to buy the good at every price. Similarly, an exogenous increase in average income would also lead the demand curve to shift outward or inward, depending on whether people choose to buy more or less of a good at a given price.

Economic Impacts of a Control Program – Multiple Markets

The above description is typical of the expected market effects for a single product market considered in isolation (for example, the ocean transportation service market). However, the markets considered in this EIA are more complicated because they are linked: the market for engines is affected by the market for vessels, which is affected by the market for international marine transportation services. In particular, it is reasonable to assume that the input-output relationship between the marine diesel engines and vessels is strictly fixed and that the demand for engines varies directly with the demand for vessels. Similarly, the demand for vessels varies directly with the demand for marine transportation services. A demand curve specified in terms of its downstream consumption is referred to as a derived demand curve. Figure 6A-5 illustrates how a derived demand curve is identified.

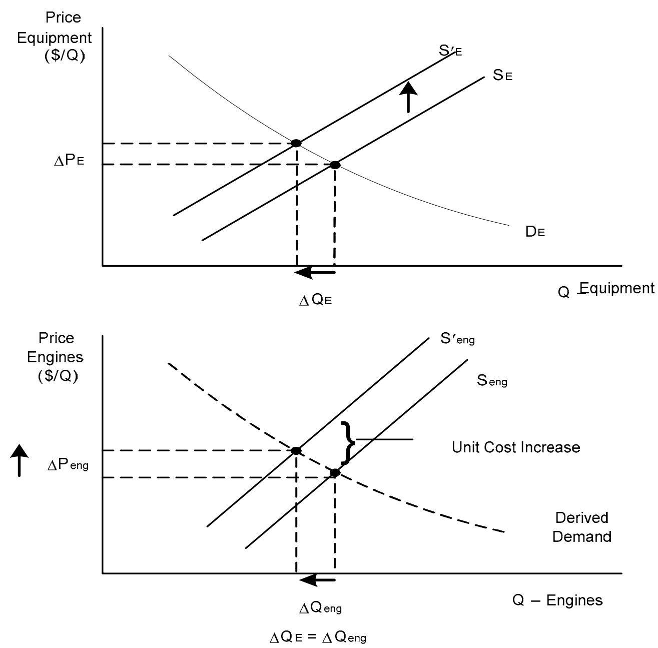

Figure 6A-5 Derived-Demand Curve for Engines

[K] An accessible detailed discussion of these concepts can be found in chapters 5-7 of Nicholson's (1998) intermediate microeconomics textbook.

Consider an event in the engine market, such as a new technology requirement, that causes the price of an engine to increase by ΔP_{eng}. This increase in the price of an engine will cause the supply curve in the engine market to shift up, leading to a decreased quantity (ΔQ_{eng}). The change in engine production leads to a decrease in the demand for equipment (ΔQ_E). The difference between the supply curves in the equipment market, $S'_E - S_E$, is the difference in price in the engine market, ΔP_{eng}, at each quantity. Note that the supply and demand curves in the equipment market are needed to identify the derived demand in the engine market.

In the market for vessels and engines, the derived demand curves are expected to be vertical. The full costs of the engines will be passed into the cost of vessels, and the cost of vessels will be passed into the cost of ocean transportation.

Using Economic Theory to Estimate the Social Costs of a Control Program

The economic welfare implications of the market price and output changes with the regulation can be examined by calculating consumer and producer net "surplus" changes associated with these adjustments. This is a measure of the negative impact of an environmental policy change and is commonly referred to as the "social cost" of a regulation. It is important to emphasize that this measure does not include the benefits that occur outside of the market, that is, the value of the reduced levels of air pollution with the regulation. Including this benefit will reduce the net cost of the regulation and even make it positive.

The demand and supply curves that are used to project market price and quantity impacts can be used to estimate the change in consumer, producer, and total surplus or social cost of the regulation (see Figure 6A-6).

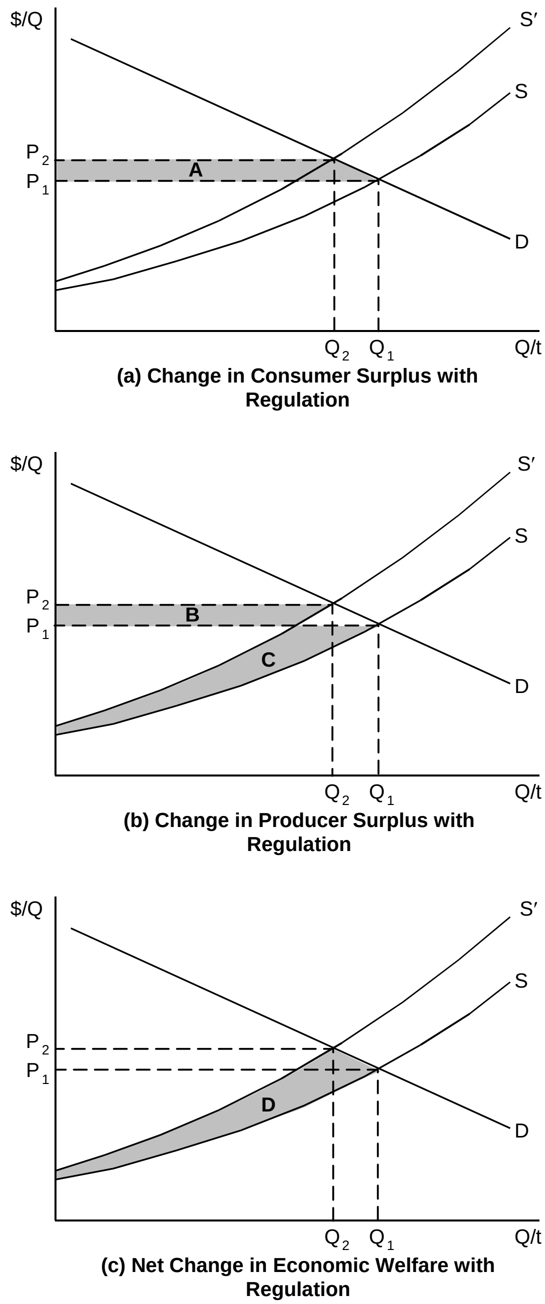

Figure 6A-6 Economic Welfare Calculations: Changes in Consumer, Producer, and Total Surplus

The difference between the maximum price consumers are willing to pay for a good and the price they actually pay is referred to as "consumer surplus." Consumer surplus is measured as the area under the demand curve and above the price of the product. Similarly, the difference between the minimum price producers are willing to accept for a good and the price they actually receive is referred to as "producer surplus." Producer surplus is measured as the area above the supply curve below the price of the product. These areas can be thought of as consumers' net benefits of consumption and producers' net benefits of production, respectively.

In Figure 6A-6, baseline equilibrium occurs at the intersection of the demand curve, D, and supply curve, S. Price is P_l with quantity Q_l. The increased cost of production with the regulation will cause the market supply curve to shift upward to S′. The new equilibrium price of the product is P_2. With a higher price for the product there is less consumer welfare, all else being unchanged. In Figure 6A-6(a), area A represents the dollar value of the annual net loss in consumer welfare associated with the increased price. The rectangular portion represents the loss in consumer surplus on the quantity still consumed due to the price increase, Q_2, while the triangular area represents the foregone surplus resulting from the reduced quantity consumed, $Q_l - Q_2$.

In addition to the changes in consumers' welfare, there are also changes in producers' welfare with the regulatory action. With the increase in market price, producers receive higher revenues on the quantity still purchased, Q_2. In Figure 6A-6(b), area B represents the increase in revenues due to this increase in price. The difference in the area under the supply curve up to the original market price, area C, measures the loss in producer surplus, which includes the loss associated with the quantity no longer produced. The net change in producers' welfare is represented by area B – C.

The change in economic welfare attributable to the compliance costs of the regulations is the sum of consumer and producer surplus changes, that is, –(A) + (B–C). Figure 6A-6(c) shows the net (negative) change in economic welfare associated with the regulation as area D.

How the Economic Theory Applied in This EIA

In the above explanation of how to estimate the market and social welfare impacts of a control action, the price elasticities of supply and demand were nonzero. This was reflected in the upward-slope of the supply curve and the downward slope of the demand curve. In the derived demand analysis, a nonzero price elasticity of demand in the vessel market yielded a nonzero price elasticity of demand in the engine market.

However, the price elasticity of demand in the international shipping market is expected to be nearly perfectly inelastic (demand curve with near-infinite slope – a vertical demand curve). This is not to say that an increase in price has no impact on quantity demanded; rather, it means that the price increase would have to be very large before there is a noticeable change in quantity demanded.

The price elasticity of demand is expected to be near perfectly inelastic because there are no reasonable alternatives to shipping by vessel for the vast majority of products transported by sea to the United States and Canada. It is impossible to ship goods between these countries and Asia, Africa, or Europe by rail or highway. Transportation of goods between these countries and Central and South America by rail or highway would be inefficient due to the time and costs involved. As a result, over 90% of the world's traded goods are currently transported by sea.[12] While aviation may be an alternative for some goods, it is impossible for goods shipped in bulk or goods shipped in large quantities. There are also capacity constraints associated with trans-continental aviation transportation, and the costs are higher on a per tonne basis.

A nearly perfectly inelastic price elasticity of demand simplifies the analysis described above. Figure 6A-7 reproduces the relationships in a multi-level market but this time with a nearly perfectly inelastic demand curve in the international shipping market. The relationships between this market and the markets for vessels and engines means that the derived demand curves for engines and vessels are also expected to be nearly perfectly inelastic. Specifically, if demand for transportation services is not expected to be affected by a change in price, then the demand for vessels will also remain constant, as will the demand for engines.

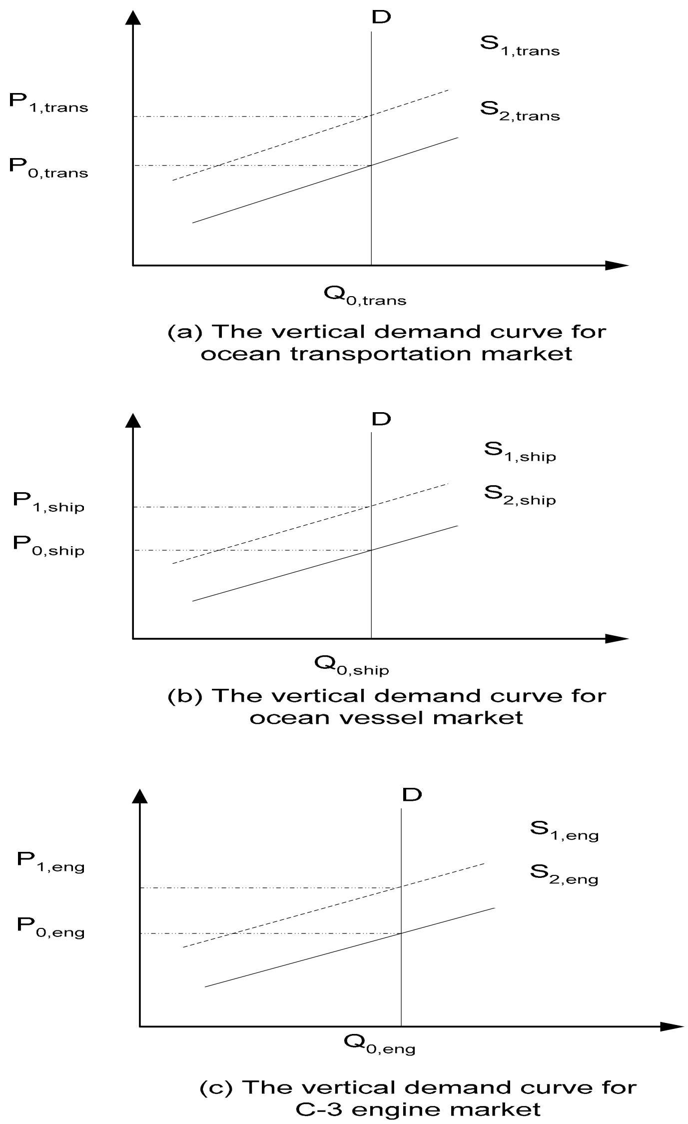

Figure 6A-7 Market Impacts in Markets with Nearly Perfectly Inelastic Demand

As indicated in Figure 6A-7, a change in unit production costs due to compliance with the engine emission and fuel sulfur requirements in the proposed ECA shifts the supply curves for engines, vessels, and ocean transportation services. The cost increase causes the market price to increase by the *full* amount of per unit control cost (i.e. from P_0 to P_1) while the quantity demanded for engines, vessels, and transportation services remains constant. Thus, engine manufacturers are expected to be able to pass on the full cost of producing Tier III compliant engines to the vessel builders, who are expected to be able to pass the full cost of installing the engines and fuel switching equipment on to the vessel owners. The vessel owners, in turn, are expected to be able to pass on these cost increases, as well as the additional operating costs they incur for the use of SCR reductant (urea) and low sulfur fuel while operating in the ECA.

Note that the fuel and urea costs affect the ocean transportation services market directly, but affect the vessel and engine markets only through the derived demand curves. That is, the equilibrium prices and quantities for vessels and engines will change only if the quantity of ocean transportation services demanded changes due to fuel and urea costs. Because the changes in fuel and urea prices are expected to be too small to affect the quantity of ocean transportation services demanded, the markets for vessels and engines are not expected to be affected by fuel changes.

The sole exception for the assumption of nearly perfectly price elasticity of demand is the cruise market. Clearly, the consumers in that market, tourists and holiday-makers, have alternatives available for their recreational activities. If the cost of a cruise increases too much, they may decide to spend their vacation in other activities closer to home, or may elect to fly somewhere instead. As a result, the costs of compliance for the cruise industry are more likely to be shared among stakeholders. If the price elasticity of demand is larger (in absolute value) than the price elasticity of supply, ship owners will bear a larger share of the costs of the program; if the price elasticity of demand is smaller (in absolute value) than the price elasticity of supply, consumers will bear a larger share of the program. Similarly, the vessel builders and engine manufacturers will also bear a portion of the costs. If the quantity demanded for cruises decreases, the derived quantity demanded for vessels will decrease, as will the derived quantity demanded for engines. If the supply curves for these industries are not perfectly elastic (i.e., horizontal), then the downward-sloping derived demand curves will lead to shared impacts among the sectors.

As described in section 6.3.3 of this chapter, the impacts on the cruise market are expected to be small, with total engine and vessel costs increasing about one percent and operating costs increasing between 1.5 and 6 percent. These increases are within the range of historic variations in bunker fuel prices. The impact on the cruise market, then, may be similar in effect to the market's response to those changes.

Finally, it may be possible for cruise ships to offset some of these costs by advertising the environmental benefits of using engines and fuels that comply with the ECA requirements. Many cruise passengers enjoy this form of recreational because it allows them a personal-level experience with the marine environment, and they may be willing to pay an increased fee to protect that nature. If people prefer more environmentally friendly cruises, then the demand curve for these cruises will shift up. Consumers will be willing to bear more of the costs of the changes. If the demand shift for environmentally friendly cruises is large enough, both the equilibrium price and quantity of cruises might increase.

Appendix 6B

Estimation of Transportation Market Impacts

The U.S. and Canada have submitted a joint proposal to IMO to designate an emission control area in which ships would need to comply with stringent fuel sulfur limits and Tier III NO_X standards. To characterize the increase in vessel operating costs due to the proposed ECA, and therefore the impacts on transportation market prices, calculations were performed for three types of ocean going vessels, container, bulk carrier, and cruise liner. Our estimates were developed using typical vessel characteristics, projected fuel and urea costs, and worst case sea-route data. This appendix presents the methodology used for these calculations.

Container Vessel

A typical container vessel was derived using data obtained from the Lloyd's of London Sea-Web Database. This data base includes information on actual vessel size (Dead Weight Tonnes (DWT)) and engine power (kilowatt – hour (kW-hr)) for a wide range of vessel types.

Operating costs included those associated with switching from residual fuel to 0.1% sulfur distillate fuel and urea consumption for vessels equipped with selective catalytic reduction (SCR). The fuel and urea costs are based on projections that are presented in the ECA proposal. These fuel costs estimates are $322/tonne for residual fuel and $468/tonne for 0.1% sulfur distillate fuel. We use a urea consumption rate of 7.5% of fuel consumption, at $1.52/gallon.

To develop a representative sea-route for our price estimations, we created a 'circle route' for a theoretical trip. Since the Port of Los Angeles[13], one of the largest ports in the U.S., lists the majority of its cargo as traveling from South Asia, our route had a vessel hypothetically travel from Singapore to the Port of Seattle, then down the West Coast of the United States (U.S.) to the Port of Los Angeles, then back to Singapore. To map this route, we divided it into three "legs." The first leg has the vessel traveling from Singapore to the Port of Seattle; the second part travels down the West Coast of the U.S. to the Port of Los Angeles/Long Beach (POLA/LB); the third leg continues from Los Angeles to Singapore. The total distance for this route was determined from http://nauticaldistance.com/, and is described below.

We understand that it will take some additional time and distance to switch vessel operations from one fuel to another. Additionally, we acknowledge that vessels may enter the ECA at an angle relative to the port in question, and would be operating in the ECA for a slightly longer distance than the 200 nautical miles of the ECA. Therefore, to make our fuel usage estimates as accurate as possible, we included some additional ECA traversing distances in our circle route calculations, adding 183 nm to the distance for reaching the Port of Seattle, and 35 nm to the distance from POLA/LB.

Baseline Operating Costs

In order to begin our estimated fuel cost increases, we needed to establish the fuel usage and prices for our baseline route (i.e. the price of the route operating on residual fuel). We determined average operational values for our hypothetical vessel by selecting the mid-point of the operational ranges used today by OGV. Therefore, our baseline estimations for the fuel usage for the first leg were determined by multiplying the engine power for the average sized containership (in kilowatts (kW)) by the average estimated engine efficiency (80 percent) as well as the average residual fuel consumption (195 grams fuel per kilowatt hour (g/kW-hr)). (Equation 6B-1) This value was then multiplied by the nautical miles (nm) for the first leg of the trip (the distance from Singapore to Seattle (7,064 nm)), and divided by the average engine speed (16 knots). To obtain the correct units for the calculation, a unit conversion was also included. (Equation 6B-2) As average values are represented here, it is possible that these values could fluctuate slightly depending on the vessel's speed, engine efficiency, and specific fuel consumption, but we believe that these estimates provide a reasonable forecast for the majority of container vessels in operation today.

Equation 6B-1

$$36{,}540kW \times 0.8 \times 195\, {}^{g_{resid}}\!/\!_{kW-hr} = 5{,}700{,}240\, {}^{g_{resid}}\!/\!_{hr}$$

Equation 6B-2

$$\frac{5{,}700{,}240\, {}^{g_{resid}}\!/\!_{hr} \times 7{,}064nm}{16\, {}^{knots}\!/\!_{hr}} \times \frac{tonne}{1{,}000{,}000g} = 2{,}517tonne_{resid}$$

The same determinations were conducted for the second leg of the trip (1,143 nm, Equation 6B-3) and the third leg (7,669 nm, Equation 6B-4).

Equation 6B-3

$$\frac{5{,}700{,}240\, {}^{g_{resid}}\!/\!_{hr} \times 1{,}143nm}{16\, {}^{knots}\!/\!_{hr}} \times \frac{tonne}{1{,}000{,}000g} = 407tonne_{resid}$$

Equation 6B-4

$$\frac{5{,}700{,}240\, {}^{g_{resid}}\!/\!_{hr} \times 7{,}669nm}{16\, {}^{knots}\!/\!_{hr}} \times \frac{tonne}{1{,}000{,}000g} = 2{,}732tonne_{resid}$$

Total fuel usage for each leg of the trip was multiplied by the price of the fuel (2006 U.S. dollars per tonne ($/tonne) which provided the baseline cost of fuel for each leg. These costs were then summed to produce an aggregate estimation of fuel cost for the entire circle trip (Equation 6B-5). This calculation provides the baseline cost of about $1.8M for an average sized container ship to traverse the theoretical circle route.

Equation 6B-5

$$(2{,}517tonne_{resid} + 407tonne_{resid} + 2{,}732tonne_{resid}) \times \$322.48 / tonne_{resid} = \$1{,}823{,}947$$

Operating Costs with an ECA

Operating cost increases due to an ECA are due to increased fuel costs and urea consumption within the ECA. Operating costs are assumed to remain unchanged outside the ECA. In addition, the ECA is assumed to have no impact on the route travelled.

Increased Fuel Costs

To determine the fuel usage and price increase caused by the ECA on our vessel traveling our theoretical circle route, we conducted the same analysis as our baseline using the appropriate distillate fuel properties. Since the distillate fuel will most likely only be used in the ECA, the remainder of the trip will continue operating on residual fuel. Therefore, we adjusted our trip section distances accordingly, using residual fuel over the first leg for 6,679 nm and over 7,434 nm for the third leg, while the remainder of the trip was determined using a distillate fuel. Equation 6B-6 provides the approximation for engine power and fuel consumption using distillate fuel and Equation 6B-7, 8, and 9 calculate the corresponding trip segment fuel usages. Due to the chemical properties of the two marine fuels, there is approximately a five percent (5%) increase in energy, on a mass basis, when operating on the distillate fuel instead of the residual fuel, and this increase is accounted for in Equation 6B-6.

Equation 6B-6

$$36{,}540kW \times 0.8 \times \frac{195\, g_{distil}/kW-hr}{1+0.05} = 5{,}428{,}800\, g_{distil}/hr$$

Equation 6B-7a Residual Fuel Estimation

$$\frac{5{,}700{,}240\, g_{resid}/hr \times 6{,}679nm}{16\, knots/hr} \times \frac{tonne}{1{,}000{,}000g} = 2{,}379 tonne_{resid}$$

Equation 6B-7b Distillate Fuel Estimation

$$\frac{5{,}428{,}800\, g_{distil}/hr \times 385nm}{16\, knots/hr} \times \frac{tonne}{1{,}000{,}000g} = 131 tonne_{distil}$$

Equation 6B-8

$$\frac{5{,}428{,}800\, g_{distil}/hr \times 1{,}143nm}{16\, knots/hr} \times \frac{tonne}{1{,}000{,}000g} = 388 tonne_{distil}$$

Equation 6B-9a Residual Fuel Estimation

$$\frac{5{,}700{,}240\, g_{resid}/hr \times 7{,}434knots}{16\, knots/hr} \times \frac{tonne}{1{,}000{,}000g} = 2{,}648 tonne_{resid}$$

Equation 6B-9b Distillate Fuel Estimation

$$\frac{5{,}428{,}800\, g_{distil}/hr \times 235nm}{16\, knots/hr} \times \frac{tonne}{1{,}000{,}000g} = 80 tonne_{distil}$$

Urea Costs

Switching to a distillate marine fuel will achieve reductions only in sulfur and particulate emissions. In order to meet the required Nitrogen Oxides (NO_X) emission reductions, vessel owners/operators would need to install a Selective Catalytic Reduction (SCR) device, or similar technologies, on new vessels built in 2016 and later. Using an SCR requires dosing exhaust gases with urea to aid with the emission reductions – which adds some additional costs to the operation of the vessel. In an SCR on a marine engine, the average dosage of urea is seven and a half percent (7.5%) per gallon of distillate fuel used. Subsequently, to estimate the volume of urea required for our circle route, we multiplied the distillate quantity determined above by this urea percentage. (Equation 6B-10) As we expect these costs to be incurred several years in the future, we used the analysis preformed for the EPA by EnSys[14] which predicted that in 2020, 33.2% of the fuel used in ECAs will be on vessels equipped SCR. The urea costs below are adjusted to reflect this prediction.

Equation 6B-10

$$599 tonnes_{distil} \times \frac{kg}{0.001 tonne} \times \frac{m^3}{836.6 kg_{distil}} \times \frac{264.17 gal}{m^3} \times 0.075 = 14{,}185 gal_{urea} \times 0.332 = 4{,}709 gal_{urea}$$

To determine the additional price of our vessel's operation through the ECA, we then multiplied the fuel and urea quantities by their corresponding prices ($322.48/tonne for residual, $467.92/tonne for distillate, and $1.52/gal for the urea). We then summed these values to determine the aggregate price for fuel and urea required for our container vessel to travel our circle route with the proposed ECA in place (Equation 6B-11).

Equation 6B-11

$$\begin{aligned}&[(2{,}379 tonne_{resid} + 2{,}648 tonne_{resid}) \times \$322.48 / tonne_{resid}] + \\ &[(131 tonne_{distil} + 388 tonne_{distil} + 80 tonne_{distil}) \times \$467.92 / tonne_{distil}] + \\ &(4{,}709 gal_{urea} \times \$1.52 / gal_{urea}) = \$1{,}908{,}549_{ECA}\end{aligned}$$

The total estimated price for an average sized containership traversing the circle with the ECA in place is just over $1.9M. The cost increase of this trip caused by the fuel and urea prices used in the ECA came from subtracting the baseline (residual fuel) trip price from the ECA price (Equation 6B-12). The price differential between the baseline trip and the ECA trip is demonstrated in Equation 6B-13 and takes into consideration the fuel cost portion of the operational cost for a vessel, which is typically around 60 percent of the total. As can be seen, by operating in the ECA for our theoretical circle route it is estimated that the operational costs due to the distillate fuel is approximately three percent (3%).

Equation 6B-12

$$\$1{,}908{,}549_{ECA} - \$1{,}823{,}947_{baseline} = \$84{,}602$$

Equation 6B-13

$$0.60 \times \frac{\$1{,}908{,}549_{ECA} - \$1{,}823{,}947_{baseline}}{\$1{,}823{,}947_{baseline}} \times 100 = 2.8\%$$

To put this price increase in some perspective, we assumed our average sized containership was hauling goods, such Twenty-foot Equivalent Units (TEU), and estimated the increase per each TEU. Estimating these prices required the cargo weight of the vessel. Literature shows that approximately 93-97% of a container vessel's DWT is used for hauling cargo, with the remaining weight composing the crew, vessel engines and hull, and fuel[15]. Equation 6B-14 shows the calculation used to convert the vessel's DWT to cargo weight using the middle value of 95%.

Equation 6B-14

$$50{,}814 DWT \times 0.95 = 48{,}273 cargo\_tonnes$$

Dividing the difference between the baseline fuel price and the ECA fuel price we calculated previously by the cargo tonnes as established in Equation 6B-14 provided the price increase per tonne of good shipped for the entire route (Equation 6B-15).

Equation 6B-15

$$\frac{(\$1{,}908{,}549_{ECA} - \$1{,}823{,}947_{baseline})}{48{,}273 cargo\_tonnes} = \$1.75 / cargo\_tonne_{increase}$$

Using this value and the weight of a full TEU (10 metric tonnes)[16], we determined the cost increase for shipping a fully loaded TEU across our circle route (Equation 6B-16).

Equation 6B-16

$$\frac{\$1.75}{cargo\_tonne_{increase}} \times \frac{10 tonnes}{full\_TEU} = \$17.53 / full\_TEU_{increase}$$

Bulk Carrier

Since the majority of goods transported to the U.S. are brought by bulk carriers as well as container vessels, and bulk carriers are of a different construction than container vessels, we also conducted estimations as to what the price increase per tonne of bulk cargo would be due to the ECA. For a comparison, we calculated what the price increase would be for a tonne of bulk cargo carried on a vessel traversing the same theoretical circle route as the containership.

Equation 6B-17 shows the same calculations as performed above for the containership using the average engine power for a bulk carrier (3,825 kW) and the total trip distance (15,876 nm)

Equation 6B-17

$$\frac{3{,}825 kW \times 0.8 \times 195 \, {}^{g_{resid}}\!/_{kW-hr} \times 15{,}876 nm}{16 \, {}^{knots}\!/_{hr}} \times \frac{tonne}{1{,}000{,}000 g} = 592 tonne_{resid}$$

This determination was also conducted for the ECA, using the appropriate values for the distillate part of the circle route (1,763 nm) and the residual fuel part of the route (14,113 nm) (Equation 6B-18 and 19 respectively). Equation 6B-20 determines the urea required for use in the ECA (as was established in Equation 6B-10), and Equation 6B-21 estimates the overall price increase for the bulk carrier if it was to operate on the theoretical circle route through the ECA.

Equation 6B-18

$$3{,}825kW \times 0.8 \times \frac{195\, {}^{g_{resid}}\!/\!{}_{kW-hr}}{1+0.05} \times \frac{1{,}763nm}{16\, {}^{knots}\!/\!{}_{hr}} \times \frac{tonne}{1{,}000{,}000g} = 62.6tonne_{distil}$$

Equation 6B-19

$$\frac{3{,}825kW \times 0.8 \times 195\, {}^{g_{resid}}\!/\!{}_{kW-hr} \times 14{,}113nm}{16\, {}^{knots}\!/\!{}_{hr}} \times \frac{tonne}{1{,}000{,}000g} = 526tonne_{resid}$$

Equation 6B-20

$$62.6tonnes_{distil} \times \frac{kg}{0.001tonne} \times \frac{m^3}{836.6kg_{distil}} \times \frac{264.17gal}{m^3} \times 0.075 = 1{,}483gal_{urea} \times 0.332 = 492gal_{urea}$$

Equation 6B-21

$$[(62.6tonne_{distil} \times \$467.92 / tonne_{distil}) + (526tonne_{resid} \times \$322.48 / tonne_{resid}) + (492gal_{urea} \times \$1.52 / gal_{urea})]$$
$$-[592tonne_{resid} \times \$322.48 / tonne_{resid}] = \$8{,}756_{increase}$$

To establish this price increase in terms of bulk cargo shipped, the value from Equation 6B-21 was divided by the available cargo weight for the bulk carrier which was determined from the actual vessel weight (16,600 tonnes) as was performed in Equation 6B-14. (Equation 6B-22)

Equation 6B-22

$$\frac{\$8{,}756_{increase}}{(16{,}600bulk\_cargo\_tonnes \times 0.95)} = \$0.56 / bulk\_cargo\_tonne_{increase}$$

As can be seen, for an average bulk carrier that would travel from Singapore to Seattle, LA/LB, and then back out to Singapore, the price increase caused by operation in the ECA would be around $0.56 per tonne of good shipped. As with the other vessels, this price would fluctuate depending on the distance traveled within the ECA, the vessel's speed, and the engine power used.

Cruise Ship

We also conducted an analysis on a typical Alaskan cruise liner. These vessels tend to operate close to shore and would be within the ECA for the majority of their routes. As such, this analysis presents worst case cost impacts for this type of vessel.

To conduct this analysis, a series of average vessel characteristics were chosen along with a typical 7 day Alaskan cruise route. The characteristics used below are the main engine power (31,500 kW), auxiliary engine power (18,680 kW), base specific residual fuel consumption (178 g_{fuel}/kW-hr for main engines, 188 g_{fuel}/kW-hr for auxiliary engines), distance between voyage destinations (5 destinations with a distance ranging between 230 to 700 nm), maximum vessel speed (21.5 knots), and the average number of passengers on-board the vessel (1,886 people).

Additionally, the arrival and departure times at the various ports of call along the cruise route were used to calculate the average speed travelled between each destination. The required power for a given journey segment was calculated using the relationship shown in Equation 6B-23. This relationship was developed for the "2005-2006 BC Ocean-Going Vessel Emissions Inventory"[17] and was shared with several cruise ship operators for their input and validation.

Equation 6B-23

$$\text{Required engine power} = 0.8199 \times (\text{avg speed/max speed})^3 - 0.0191 \times (\text{avg speed/max speed})^2$$
$$+ 0.0297 \times (\text{avg speed/max speed}) + 0.1682$$

This relationship was developed to approximate effective power given cruise ships' diesel-electric operation. The auxiliary engines reported within the Lloyd's of London 'Seaweb' database[18], and are presumably operated independently of the vessels main diesel-electric power generation, as well as assumed to operate at an average of 50% power for the entire voyage.

To demonstrate the price increase for the cruise liner that would operate within the ECA, calculations for one leg of the Alaskan voyage are shown in Equation 6B-24-27, the entire trip operational cost increase per person in Equation 6B-28, and with Table 6B-1 depicting the total increases over the entire trip broken out by destination.

Equation 6B-24

$$31{,}500kW \times 0.5683 \times \frac{178g_{fuel}}{kW-hr} \times 704knots \times \frac{hr}{16.76knots} \times \frac{tonne}{1{,}000{,}000g} = 134tonne_{resid}$$

Equation 6B-25

$$\frac{134tonne_{resid}}{1{,}886\,people} \times \frac{\$322.48}{tonne_{resid}} = \$22.89 / person_{resid}$$

Equation 6B-26

$$31{,}500kW \times 0.5683 \times \frac{178g_{fuel}}{(1.05)kW-hr} \times 704knots \times \frac{hr}{16.76knots} \times \frac{tonne}{1{,}000{,}000g} = 127tonne_{distil}$$

Equation 6B-27

$$\frac{127tonne_{distil}}{1{,}886\,people} \times \frac{\$467.92}{tonne_{distil}} = \$31.62 / person_{distil}$$

Equation 6B-28

$$\$31.62 - \$22.89 = \$8.73 / person_{main\_increase}$$

Table 6B-1 Alaskan Cruise Liner Destinations and the Corresponding Operational Price Increases

DESTINATION ORIGIN	DESTINATION CONCLUSION	ESTIMATED PRICE INCREASE / PERSON ($)
Vancouver	Sitka	$8.73
Sitka	Hubbard Glacier	$3.06
Hubbard Glacier	Juneau	$2.67
Juneau	Ketchilkan	$2.42
Ketchilkan	Vancouver	$6.13
Total		**23.02_{main_increase}$**

Additionally, the operational cost increases for the auxiliary engines were estimated (Equation 6B-29-33), as well as the cost increases caused by dosing the engine exhaust with urea (Equation 6B-34& 35), and the total price increase for the cruise (Equation 6B-36) divided by the length of the cruise (Equation 6B-37).

Equation 6B-29

$$18{,}680kW \times 0.50 \times \frac{188g_{fuel}}{kW - hr} \times 168hrs \times \frac{tonne}{1{,}000{,}000g} = 295tonne_{resid}$$

Equation 6B-30

$$\frac{295tonne_{resid}}{1{,}886\,people} \times \frac{\$322.48}{tonne_{resid}} = \$50.44 / person_{resid}$$

Equation 6B-31

$$18{,}680kW \times 0.50 \times \frac{188g_{fuel}}{(1.05)kW - hr} \times 168hrs \times \frac{tonne}{1{,}000{,}000g} = 281tonne_{distil}$$

Equation 6B-32

$$\frac{281tonne_{distil}}{1{,}886\,people} \times \frac{\$467.92}{tonne_{distil}} = \$69.71 / person_{distil}$$

Equation 6B 33

$$\$69.71 - \$50.44 = \$19.27 / person_{aux\_increase}$$

Equation 6B-34

$$616.75tonnes_{distil} \times \frac{kg}{0.001tonne} \times \frac{m^3}{836.6kg_{distil}} \times \frac{264.17gal}{m^3} \times 0.075 = 14{,}606gal_{urea} \times 0.332 = 4{,}849gal_{urea}$$

Equation 6B-35

$$\frac{4{,}849gal_{urea}}{1{,}886\,people} \times \$1.52 / gal_{urea} = \$3.91_{urea\_increase}$$

Equation 6B-36 $$\$23.02_{main\_increase} + \$19.27_{aux\_increase} + \$3.91_{urea\_increase} = \$46.20 / person_{total\_increase}$$

Equation 6B-37

$$\frac{\$46.20 / person_{total\_increase}}{7days_{cruise\_length}} = \$6.60 / person / day$$

To put this price increase in perspective of the additional cost for a typical seven-day Alaskan cruise, we also determined the % increase for the various stateroom types available on the vessel. These values were established as shown in Equation 6B-38 and Table 6B-2 lists the four main stateroom types used on a typical Alaskan cruise liner.

Equation 6B-38

$$\frac{\$46.20}{Stateroom\_price(\$599)} \times 100 = 7.7\%$$

Table 6B-2 Representative Alaskan Cruise Liner Stateroom Price Increases

STATEROOM TYPE	ORIGINAL AVERAGE PRICE PER NIGHT ($)	PERCENTAGE INCREASE
Interior	$100	6.6%
Ocean View	$200	3.3%
Balcony	$300	2.2%
Suite	$400	1.7%

As can be seen from all the above price increase estimations, the additional costs of the distillate fuel and the urea required to operate in the proposed ECA will not be a significant monetary increase to the overall operation of the vessel, regardless of vessel type.

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