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Table of Contents

List of Tables

List of Figures

Acronyms and Abbreviations

AGNPS – Agricultural Nonpoint Source Model

ASCE – American Society of Civil Engineers

ATS – Active Treatment Systems

ATTAINS – Assessment TMDL Tracking and Implementation System

AWWA – American Water Works Association

BMP – Best Management Practices

BOD – Biological Oxygen Demand

BUVD – Benefits Use Valuation Database

C&D – Construction and Development

cfs – cubic feet per second

ChA – Chlorophyll-a

cm - centimeter

CPI – Consumer Price Index

CS – Compensating Surplus

CWA – Clean Water Act

DAF – Dissolved Air Flotation

DCPs – Dissolved Concentration Potentials

DEMs – Digital Elevation Models

DO – Dissolved Oxygen

EDA – Estuarine Drainage Area

EMAP-NCA – Environmental Monitoring & Assessment Program, National Coastal Assessment Monitoring Data

EPA – Environmental Protection Agency

EPT – Ephemeroptera, Plectoptera, and Trichoptera

ERF – Enhanced Reach File

EVRI – Environmental Valuation Resource Inventory

FC – Fecal Coliform

ft/s – feet per second

FTUs – Formazin Turbidity Units

g - Gram

GIS – Geographic Information System

HAB – Harmful Algal Bloom

HARC – The Houston Area Research Center

HSPF – Hydrologic Simulation Program-Fortran

HUC – Hydrologic Unit Code

JTU – Jackson Turbidity Units

kg/ha – Kilograms per hectare

LC – Lethal Concentration

LID – Low Impact Development

m - Meter

MCL – Maximum Contamination Limit

mgd – Millions of gallons per day

μg/L – Micrograms per liter

mg/L – Milligrams per liter

MIB – 2-methylisoborneol

mL/L – Milliliters per liter

mm – Millimeter

MRLC – Multi-Resolution Land Characteristics Consortium

NAWQA – National Water-Quality Assessment

NBER – National Bureau of Economic Research

NCEE – National Center for Environmental Economics

NEIWPCC – New England Interstate Water Pollution Control Commission

NERRS – The National Estuarine Research Reserve System

NHD – National Hydrogeography Dataset

NID – National Inventory of Dams

NLCD – National Land Cover Database

NMFS – National Marine Fisheries Service

NOAA – National Oceanic and Atmospheric Administration

NPDES – National Pollutant Discharge Elimination System

NRC – National Research Council

NRCS – Natural Resource Conservation Service

NRI – National Resources Inventory

NSQAN – National Stream Quality Accounting Network

NTUs – Nephelometric Turbidity Units

NWIS – National Water Information System

NWLS – Nonlinear Weighted Least Squares

PAC – Powdered Activated Carbon

PAHs – Polycyclic Aromatic Hydrocarbons

POTWs – Publicly Owned Treatment Works

PPI – Producer Price Index

PTS – Passive Treatment Systems

RESIS – Reservoir Sedimentation Survey Information System

RF1 – Reach File Version 1.0

RFF – Resources for the Future

RMSE – Root Mean Squared Error

RR – River Reach

RUSLE – Revised Universal Soil Loss Equation

RUSLE k-factor – Soil Erodibility

RUSLE r-factor – Precipitation

SABS - Suspended and Bedded Sediments

SAS – Statistical Analysis System

SAS IML – Statistical Analysis System Interactive Matrix Language

SAV – Submerged Aquatic Vegetation

SCDHEC – South Carolina Department of Health and Environmental Control

SCDNR – South Carolina Department of Natural Resources

SDWIS – Safe Drinking Water Information System

SPARROW – Spatially Referenced Regressions on Watershed Attributes

SSC – Suspended Sediment Concentration

STATSGO - State Soil Geographic database

STORET – Storage and Retrieval data warehouse

SWAT – Soil Water Assessment Tool

T&E – Threatened and Endangered

TDS – Total Dissolved Solids

TESS – Threatened and Endangered Species System database

TMDL – Total Maximum Daily Load

TM/ETM – Thematic Mapper/Embedded Trace Macrocells

TN – Total Nitrogen

TP – Total Phosphorus

TSS – Total Suspended Solids

U.S. OMB – United States Office of Management and Budget

USACE – United States Army Corps of Engineers

USDA – United Stated Department of Agriculture

EPA – United States Environmental Protection Agency

USFWS – United States Fish and Wildlife Service

USGS – United States Geological Survey

USLE – Universal Soil Loss Equation

WATERS – Watershed Assessment Tracking & Environmental Results

WHO – World Health Organization

WQI – Water Quality Index

WQL – Water Quality Ladder

WSUT – Weak Structural Utility Theoretic

WTP – Willingness To Pay

WWF – World Wildlife Fund

WY – Water Year

yd^3 – cubic yard

1 Introduction

This document presents information on environmental impacts associated with construction site discharges to surface waters and benefits associated with their reduction. Additional information on the construction industry and construction site discharges to surface waters is provided in the U.S. Environmental Protection Agency's (EPA's) *Development Document for Final Effluent Guidelines and Standards for the Construction and Development Category* (USEPA 2009b) and *Economic Analysis for Final Effluent Guidelines and Standards for the Construction and Development Category* (USEPA 2009c).

Construction takes place on approximately 853,000 acres in the coterminous United States and discharge more than 5 billion pounds of sediment each year (USEPA 2009b). All major surface water types receive construction discharges, including streams, rivers, wetlands, lakes, estuaries, and other coastal waters. In any given year, some level of construction activity occurs in the majority of U.S. watersheds. However, most construction acreage is concentrated in a relatively limited number of watersheds. Between 1992 and 2001, more than half of all construction activity took place in less than 5 percent of U.S. watersheds.[1] A common development pattern is for new construction to concentrate in rural and suburban watersheds adjacent to more densely developed urban areas. Because construction is a temporary activity, the locations of the most highly impacted watersheds in the United States shift over time.

Construction activities significantly change the surface of the land. Typical activities include clearing vegetation and excavating, moving, and compacting earth and rock. Consequences from these activities include reduced stormwater infiltration, increased runoff volume and intensity, and higher soil erosion rates.

Construction sites have been documented to increase pollutant discharges to surface waters. The most thoroughly documented pollutants are sediment and turbidity. Nitrogen and phosphorus are common soil constituents that can also pollute receiving waters. Other pollutants can derive from a wide variety of construction equipment and materials or from historic contamination of construction sites. These pollutants include metals, trash and debris, nutrients, organic matter, pesticides, petroleum hydrocarbons, polycyclic aromatic hydrocarbons (PAHs), and other toxic organics. Construction activity can also impact receiving waters by increasing the volume and intensity of stormwater runoff from a site. These flows can erode receiving water banks and beds, particularly when pipes, ditches, or other stormwater conveyances concentrate discharges. Surface water erosion can alter waterbody morphology and elevate sediment and turbidity levels downstream.

Most pollutants enter surface waters when precipitation erodes soil and carries particulate matter to receiving waters. A number of pollutants bind to soil particles and travel with them as they erode. Other pollutants dissolve in precipitation and are carried to surface waters in solution. Construction site pollutants can also enter surface waters during dry weather due to activities such as excavation dewatering, construction equipment washout, wind erosion, and equipment operation in or near surface waters.

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[1] Watersheds are those associated with the more than 62,000 surface water reaches in the coterminous United States in the Reach File Version 1 (RF1) surface water network (USEPA 2007f). See *Chapter 4* for additional information.

Increases in pollutant and stormwater discharges from construction sites and subsequent increases in waterbody pollutant levels can have adverse impacts on aquatic ecosystem function and human aquatic resource use. Construction is a temporary activity in any given location, but impacts can range from temporary to long-term. Impacts include both physical and chemical impacts on waterbodies and biological impacts on aquatic organisms and communities. Impacts to human aquatic resource uses can include impaired drinking water supplies, recreation, navigation, fishing, water storage, aesthetics, property value, irrigation, industrial water supplies, and stormwater (including flood) management. Though some surface waters can eventually recover from construction discharge impacts, they may continue to be degraded by excess stormwater and pollutants from buildings, roads, and other structures put in place by construction activity. This document, however, focuses solely on the impacts associated with active construction sites. *Chapters 2 and 3* provide additional information on the pollutants and environmental impacts associated with construction site discharge to surface waters, with *Chapter 2* focusing on sediment and turbidity and *Chapter 3* presenting all other pollutants associated with construction site stormwater discharges. *Chapter 4* presents a review of literature documenting impacts of construction site stormwater discharges.

EPA has established effluent limitations guidelines (ELGs) and new source performance standards (NSPS) for stormwater discharges from the construction and development industry. These guidelines and standards require discharges from certain construction sites to meet a numeric turbidity limit. The guidelines and standards also require all construction sites currently required to obtain a National Pollutant Discharge Elimination System (NPDES) permit to implement a variety of best management practices (BMPs) designed to limit erosion and control sediment discharges from construction sites. EPA evaluated four options in developing the final rule. These options are described below:

- Option 1 establishes minimum requirements for implementing a variety of erosion and sediment controls and pollution prevention measures on all construction sites that are required to obtain a permit.
- Option 2 contains the same requirements as Option 1. In addition, construction sites of 30 or more disturbed acres would be required to meet a numeric turbidity limit in stormwater discharges from the site. The technology basis for the numeric limit is Active Treatment Systems (ATS). The numeric turbidity standard would be applicable to stormwater discharges for all storm events up to the local 2-year, 24-hour event.
- Option 3 contains the same requirements as Option 1. Option 3 also requires all sites with 10 or more acres of disturbed land to meet a numeric turbidity standard based on the application of ATS. The turbidity standard would apply to all stormwater discharges for all storm events up to the local 2-year, 24-hour event.
- Option 4 contains the same requirements as Option 1. Option 4 also requires all sites with 10 or more acres of disturbed land to meet a numeric turbidity standard of 280 NTU based on the application of Passive Treatment Systems (PTS). The turbidity standard would apply to all stormwater discharges for all storm events up to the local 2-year, 24-hour event, although only certain types of discharges would require monitoring.

EPA expects that reduced discharges of pollutants from construction sites will enhance environmental services provided by the affected water bodies and, as a result, human welfare. *Chapter 5* provides an overview of EPA's approach to assessing and quantifying the benefits of reducing construction site discharges to surface waters. *Chapter 6* describes the methodology EPA used to quantify improvements in surface water quality associated with each of the evaluated regulatory options. EPA used the Spatially

Referenced Regressions on Watershed Attributes (SPARROW) method, the Dissolved Concentration Potential (DCP) method, and information on relationships between sediment and nutrient levels in surface waters to quantify changes in surface water levels of total suspended solids (TSS), sedimentation, total phosphorus (TP), and total nitrogen (TN) under each regulatory option. *Chapter 6* also presents the results of this analysis. EPA found available data on the location and magnitude of other types of construction site pollutant discharges to be insufficient for inclusion in the water quality modeling analysis.

In analyzing benefits of the regulation, EPA quantified and monetized economic benefits from reduced dredging of navigable waterways, reduced dredging of water storage facilities (reservoirs), and reduced drinking water treatment costs. *Chapters 7*, *8*, and *9* describe the methods EPA used to analyze these benefit categories and present the results of EPA's analysis. Other benefit categories, including reduced flood risk, increases in property values, industrial water use, agricultural water use, stormwater management system management, and commercial fishing, are discussed qualitatively in *Chapters 2* and *5*. EPA estimates that Options 1, 2, 3, and 4 will reduce expenditures on navigable waterway dredging by approximately $1.3, $2.6, $3.3, and $2.9 million per year respectively. Expenditures on reservoir dredging are expected to decrease by $1.4 million per year under Option 1, $2.9 million per year under Option 2, $3.6 million per year under Option 3, and $3.2 million per year under Option 4. Reductions in drinking water treatment costs are estimated to amount to $1.2, $1.8 million, $2.1 million, and $1.8 million annually under Options 1, 2, 3 and 4, respectively. Overall, EPA expects this regulation to save governments and private entities between $3.8 and $8.9 million in these three areas each year, depending on the policy option.

EPA also expects that reductions in pollutant discharges to surface waters resulting due to regulation will enhance or protect aquatic ecosystems. The drop in pollutant discharges is expected to improve the protection of resident species; enhance the general health of fish, invertebrate, plant and other aquatic organism populations; increase their propagation in waters currently impaired; and expand fisheries for both commercial and recreational purposes. Improvements in water quality such as decreased turbidity will also favor increased recreational activities such as swimming, boating, nature observation, fishing, camping and other outings, as well as overall aesthetic enjoyment. Improvements associated with reduced nitrogen and phosphorus discharges include reduced eutrophication of surface waters, fewer beach closings, greater fisheries productivity, and greater enjoyment of water resources. Finally, EPA expects that the regulation will augment nonuse values (values that do not depend on use by humans, see *Chapter 5* for details) of the affected water resources. EPA used a meta-analysis of surface water valuation studies to estimate the total value of nonmarket benefits (values that arise outside of market transactions, such as recreation at publicly accessible sites) stemming from the regulation. EPA estimates that mean total willingness to pay (WTP) for water quality improvements resulting from regulation ranges from $210 million to $413 million, depending on the regulatory option under consideration. *Chapter 10* of this report provides details on EPA's analysis of nonmarket benefits. EPA expects that nonmarket benefits resulting from Options 1, 2, 3, and 4 will be approximately $210, $353, $413, and $361 million per year, respectively. Combining these nonmarket benefits estimates with the avoided cost estimate produces total benefits estimates of $214 million per year for Option 1, $360 million per year for Option 2, $422 million per year for Option 3, and $369 million per year for Option 4.

Sufficient data were available to monetize benefits for only a subset of:

- pollutants discharging from construction sites
- impacted surface waters
- ecological services from surface waters.

The monetized benefits estimates therefore represent only a portion of the total benefits of each of the regulatory options. The scope of the monetized benefits analysis is discussed in more detail in *Chapter 11*.

2 Overview of Environmental Impacts from Construction Site Sediment and Turbidity Discharges

This chapter summarizes information available from the literature and other sources on the pollutants and environmental impacts associated with construction site discharges of sediment.

Construction sites have been documented to increase pollutant discharges to surface waters in a number of studies. Sediment and turbidity are the most thoroughly documented pollutants. This chapter discusses the process of sediment and turbidity discharge from construction sites, their behavior in surface waters, and their potential impacts on aquatic organisms and human use of aquatic resources. Documentation of these discharges and impacts can be found in the literature reviewed and summarized in *Chapter 4*. A number of other pollutants can also discharge from construction sites including nitrogen, phosphorus, metals, toxic organic compounds, and others. These are discussed in *Chapter 3.*

Suspended and bedded sediments (collectively referred to as “SABS” or “sediment” in this document) and turbidity are natural components of many aquatic ecosystems. Sediments contribute to the physical structure of surface waters and help to transport nutrients and organic matter. Natural turbidity can modulate levels of aquatic photosynthetic activity and predator–prey relationships. Undisturbed ecosystems contain species adapted to the sediment and turbidity levels naturally associated with those ecosystems.

At excessive levels, however, sediment and turbidity become pollutants. Modifications to the physical and chemical composition of sediment and turbidity discharges can also transform them into pollutants. Sediment and turbidity are the most commonly documented pollutants in construction site discharges and impacted surface waters. In a number of documented cases, sites have discharged sediment and turbidity at very high levels (see *Chapter 4*).

Although suspended sediment, bedded sediment, and turbidity are distinct and separate water quality properties, all describe impacts associated with eroded soil discharge to surface waters. For this reason, many studies discuss these properties concurrently, as does this document. Soil and sediment are composed of a variety of components including organic matter, phosphorus, nitrogen, metals, and other compounds, both natural and anthropogenic. Many of these components travel with soil as it erodes and discharges to surface waters. These components are discussed in more detail in *Section 2.2.*

There are multiple terms available to describe levels of sediment and turbidity in water. *Table 2-1* presents terms and definitions used in this document.

Table 2-1: Sediment and Turbidity Terminology

Sediment Metric	Description
Bedded sediment	A general term for sediment that, at any given time, settles from the water column onto surfaces within a surface water (e.g., channel bed or aquatic plant leaves) in a process commonly known as sedimentation or siltation. Measures of bedded sediments include depth of deposition within a given time period, percent fines, geometric mean diameter, Fredle number (a permeability index) (Berry et al. 2003), and others.
Settleable solids	A measure of the solids that will settle to the bottom of a cone-shaped container (called an Imhoff cone) in a 60-minute period. Settleable solids are primarily a measure of particles that can be removed from water by sedimentation. Expressed as milliliters per liter (ml/L).
Suspended sediment	A general term for sediment that, at any given time, is either maintained in suspension by a surface water's turbulent currents or that exists in suspension as a colloid.
Suspended sediment concentration (SSC)	The velocity-weighted concentration of suspended sediment in the sampled zone in a surface water defined as extending from the water's surface to a point approximately 0.3 feet above the bed. It is determined by measuring the dry weight of all sediment from a known volume of sample. Expressed as milligrams of dry sediment per liter of water-sediment mixture (mg/L).
Total suspended solids (TSS)	A dry weight measure of suspended inorganic and organic material in the water column. It is measured by filtering a subsample of water and measuring the weight of the dried solids. Expressed in milligrams of solids per liter of water-solids mixture (mg/L).
Turbidity	A measure of the scattering and absorption of light when it enters a water sample. The quantity of suspended particles in water helps to determine turbidity levels as do particle shape, size, and color distributions. Suspended particles can include clay, silt, colloids, finely divided organic and inorganic matter, soluble colored organic compounds, plankton, and other microscopic organisms. In this document, turbidity levels are typically expressed in nephelometric turbidity units (NTUs). Higher NTU levels indicate more turbid water.

Surface water turbidity levels are controlled by the quantity and nature of particulate matter suspended in the water column. This particulate matter can consist of mineral particles (sediment), algae and other organisms, and organic detritus. Particle shape, size, and color distributions influence total turbidity levels as well. Suspended sediment contributes to surface water turbidity. However, because several factors beyond the mass of suspended solids in the water column control surface water turbidity, the quantitative relationship between suspended particle concentrations and turbidity levels has been found to vary among watersheds, surface waters, and precipitation events.

The sections below provide additional information on the nature of construction site sediment and turbidity discharges to surface waters (*Section 2.1*), their behavior and transport in surface waters (*Section 2.2*), their impacts on aquatic ecosystems (*Section 2.3*), their impacts on human use of aquatic resources (*Section 2.4*), appropriate levels in surface waters as delineated by water quality criteria (*Section 2.5*), and the extent to which they currently impair surface waters in the United States (*Section 2.6*).

2.1 Sediment and Turbidity Discharge to Surface Waters

Construction activities increase soil vulnerability to erosion. Typical construction site activities include clearing vegetation and excavating, moving, and compacting earth and rock. Vegetation removal and surface work loosens soil, removes protective root structures, and exposes soil directly to the erosive powers of precipitation and stormwater runoff. Soil compaction reduces precipitation infiltration and increases overland water flow, thereby increasing the quantity of stormwater runoff available to erode soil. In addition, stockpiled construction materials such as stripped topsoil, fill material, and soil from foundation excavation are often placed in steep, uncovered piles vulnerable to erosion. Construction vehicles track soil onto roadways from which it can easily wash into storm sewer drainage systems and subsequently to surface waters. Susceptibility to erosion remains high at construction sites until soil-disturbing activities are complete and the land surface is revegetated or otherwise stabilized.

Precipitation events are the primary cause of construction site sediment and turbidity discharges to surface waters. Raindrop impact energy and overland water flow detach soil particles from the land surface, suspend them in surface flow, and transport them to other locations on the construction site or to discharge points. Suspended soil particles in the stormwater flow create turbidity.

Sediment erosion rates are highly variable among sites and depend on a number of factors including site topography (slope length, steepness, and shape), precipitation intensity and quantity (i.e., rainfall erosivity), soil type (particle size, erodibility, land use (vegetation cover, erosion control practices), and nature of the construction activity. Some phases of construction activity disturb soil more than others. For a more detailed discussion of sediment and turbidity discharges from construction sites, see EPA's *Development Document for Final Effluent Guidelines and Standards for the Construction and Development Category* (USEPA 2009b).

Mobilization of soil particles is dependent on many factors including soil particle size, soil cohesiveness, and rainfall energy and duration. As flows grow larger and more powerful, they are able to transport larger particles. Sand-sized and larger particles are more easily detached from the soil surface because they are generally not cohesive. However, once mobilized, they more easily settle from stormwater runoff because of their greater mass. Smaller particles, such as clays, are generally harder to mobilize because they are more likely to be cohesive. Once mobilized, however, individual smaller particles are more likely to stay suspended in stormwater flow and be transported off-site because of their lesser mass (Reed 1980). The soil fraction composed of smaller particles tends to contain a disproportionate quantity of the organic matter and adsorbed materials and contaminants (e.g., metals, nutrients, pesticides, and other organic compounds) found in soils. Smaller particles also contribute disproportionately to turbidity levels.

Suspended sediment and turbidity levels in construction site stormwater flows can be very high. Suspended sediment concentrations up to 160,000 mg/L in construction site stormwater have been documented (Daniel et al. 1979; Horner et al. 1990; Warner and Collins-Camargo 2001; Hedrick et al. 2006; USEPA 2009b). Turbidity levels up to 30,000 NTU have been documented (Lubliner and Golding 2005; Bhardwaj and McLaughlin 2008). These figures reflect suspended sediment and turbidity levels prior to treatment on-site and/or prior to discharge to and dilution in surface waters.

In addition to elevated stormwater sediment and turbidity concentrations, stormwater runoff volume can also increase on construction sites (Yorke and Davis 1972; Selbig et al. 2004; Clausen 2007; Line and White 2007; Selbig and Bannerman 2008; Montgomery County DEP 2009) (see *Chapter 4*).

The path stormwater travels varies among construction sites. Stormwater can infiltrate soil, flow over land, or travel through underground storm sewer systems. Vegetated areas can provide opportunities for stormwater and pollutant capture on the land surface whereas paved surfaces tend to efficiently transport stormwater downslope. Storm sewer systems can also very efficiently move water away from construction sites (and into surface waters). Storm sewer systems are often installed early in site development when large areas of earth are still disturbed and highly prone to erosion (NRC 2008).

Distance to surface water also varies among construction sites. Some sites are far from surface waters, whereas others are adjacent to or, in some cases, directly in water (e.g., bridge construction, stream channelization). Longer travel paths prior to surface water discharge can provide more opportunities for stormwater and pollutant capture (Yorke and Herb 1978).

The combination of elevated sediment concentrations and elevated runoff volume results in higher total export of sediment from a site. Erosion is a natural process; however, sediment yields from construction sites can be many times higher than those from agricultural, forested, and mature developed sites.

Forested watersheds in the United States are estimated to yield 0.02 to 1 tons/acre/year of sediment. Cropland in the United States is estimated to yield 0.65 to 15 tons/acre/year (Vice et al. 1969; Yorke and Herb 1978; Osterkamp et al. 1998; Faucette et al. 2007). Paul and Meyer (2001) state that construction activity generally increases sediment yields 100 to 10,000 times over those from forested areas, with larger increases possible in basins with steep topography. Studies summarized in *Table 4-2* report sediment yields of 0.009 to 219 tons/acre/year from catchments containing construction. All studies of construction site sediment yield report that construction activity increases sediment discharges from sites.

Because of the importance of precipitation and its subsequent flow over land to the transport of soil, most sediment and turbidity discharges take place during or shortly after precipitation events. Once a precipitation event ceases, discharges from construction sites generally cease within a relatively short time period unless site terrain or stormwater management systems, such as stormwater ponds, attenuate discharge flows. This dynamic creates an intermittent discharge of pollutants from most construction sites.

In addition to being an episodic discharger of pollutants, construction is a temporary activity at any one site that typically lasts several months to several years. Individual construction sites are therefore transient sources of pollutant discharges to surface waters. Sediment discharges from the land surface to surface waters due to active construction at an individual site are greatly reduced once active construction ceases and site soils have been stabilized (e.g., through revegetation or paving).

Surface water impacts such as elevated turbidity and suspended sediment levels generally cease immediately downstream of construction sites soon after site discharges cease. Other impacts, however, persist beyond the lifespan of individual precipitation events, individual construction sites, or even the presence of construction activity in a watershed. This is due to the persistence of sediments and some associated pollutants in surface waters as well as longer-term impacts on aquatic organism communities.

In addition, most construction activities in the United States are concentrated in a relatively small number of watersheds during any single decade (see *Chapter 6*). Within these watersheds, the location of individual construction sites changes from year to year, but the watershed's total annual construction acreage remains elevated for a number of years until the watershed is "built out." Surface waters draining these watersheds therefore experience a persistent, elevated level of impact from construction activity for several years, despite the transient nature of individual sites in the watershed.

Sediment and turbidity discharges can also take place during dry weather at some construction sites. Dewatering of site depressions and foundation excavations, slope failure, attenuated drainage from stormwater basins, construction equipment operation or other activity directly in or near surface waters (e.g., channelization, pipeline crossing, culvert emplacement, bridge construction), vehicle and other construction equipment washing, 6ndscape irrigation, and overland flow from groundwater seeps can create discharges during dry weather. Erosion of bare ground due to snowmelt in the spring can be high (NRC 2008). Construction activity can also destabilize slopes under both dry and wet conditions. If situated sufficiently close to a waterbody, slope failure can result in mass loading of sediment and associated turbidity.

Requirements for construction site erosion and sediment control have gradually increased since the 1950s. EPA, the U.S. Federal Highway Administration, the U.S. Federal Energy and Regulatory Commission, the U.S. Department of Agriculture's Soil Conservation Service, states, counties, municipalities and other entities have issued a variety of regulations and guidance. EPA has previously issued requirements for construction sites under the National Pollutant Discharge Elimination System (NPDES) stormwater program. Phase I of the stormwater program was promulgated in 1990 and addresses large construction activities disturbing 5 or more acres. Phase II of the stormwater program was promulgated in 1999 and addresses construction sites disturbing 1 to 5 acres. EPA's *Development Document for Final Effluent Guidelines and Standards for the Construction and Development Category* (USEPA 2009b) provides additional information on the history of construction site erosion and sediment control requirements.

Under current practice, many construction site discharges are moderated to some degree by a variety of sediment erosion and stormwater discharge control practices. Practices include the use of sedimentation basins, silt fences, vegetative filter strips, grass swales, check dams, erosion control blankets, straw bale barriers, gravel bag berms, sand bag barriers, straw mulch, hydraulic mulch, wood mulch, soil binders, geotextiles and mats, rock filters, earth dikes and drainage swales, velocity dissipation devices, sedimentation traps, and other methods. These practices and others are discussed in detail in EPA's *Development Document for Final Effluent Guidelines and Standards for the Construction and Development Category* (USEPA 2009b).

2.2 Sediment and Turbidity Behavior in Surface Waters

Sediment and turbidity movement within surface waters is highly variable. Factors influencing sediment and turbidity movement and persistence in surface waters include the intensity, quantity, and composition of sediment and turbidity discharges and the nature of the receiving waters. Important waterbody characteristics include type, size, and flow rate.

Sediment discharges from construction sites typically contain a large proportion of inorganic material that can persist in surface waters for long periods of time. Under appropriate conditions (e.g., anaerobic), organic material in construction site discharges can also persist. For these reasons, impacts from construction site discharges can last beyond the life span of a single precipitation event, an individual construction site, and of the presence of construction activity in a watershed (Yorke and Herb 1978). Several researchers have stated that long-term monitoring beyond the cessation of a construction project is necessary in some cases to fully document its impacts (Barton 1977; Taylor and Roff 1986; Chen et al. 2009; Lee et al. 2009).

While in the surface water network, sediments can cycle many times between suspension in the water column, where they contribute to turbidity levels, and deposition on waterbody beds. This process depends on surface water currents and other disturbances. Larger sediment particles are more likely to settle on surface beds, and smaller particles are more likely to remain suspended in the water column and contribute to surface water turbidity. Because of this dynamic, larger particles tend to take longer to move through surface water systems than smaller particles.

Construction sites discharge sediment and turbidity to all major surface water categories: wetlands; streams and rivers; lakes, reservoirs, ponds, and other impoundments; and estuaries, bays, and other coastal waters. Some waterbodies have flow regimes and physical structures that allow them to purge or absorb excess sediment and turbidity inputs within relatively short time frames. Other surface waters allow excess sediment and turbidity to persist for long periods of time. The ability of individual surface waters to transport sediment varies as precipitation and other factors modify the nature of their flow. Surface waters also vary widely in their size relative to construction site discharges. Larger surface waters generally have more capacity for dilution of pollutant discharges. Sediment transport in several surface water types are discussed in more detail in *Section 2.2*.

Sediments and turbidity migrate with surface water flow downstream and therefore affect waters beyond the initial receiving water. Migration of construction sediment typically benefits the initial receiving water but impacts downstream waters. Several studies have documented improvements in stream conditions once construction site sediment and turbidity discharges cease and accumulated sediments migrate downstream from the initial point of entry. The sediments, however, remain within the surface water network and are redistributed to new waterbodies downstream. Because of sediment's ability to persist and migrate downstream, surface waters downstream of a watershed containing a consistent level of construction activity from year to year can experience a relatively consistent level of impact from construction sediment and turbidity discharges, even though the timing of individual precipitation events and the locations of individual construction sites within the watershed change from year to year.

2.2.1 Sediment Characteristics Affecting Surface Water Transport

Important sediment characteristics that influence transport and fate include the size, shape, density, fall velocity, and concentration of sediment particles (Shen and Julien 1993). Multiple size scales exist. *Table 2-2* presents the broad size categories of one scale.

Table 2-2: Sediment Grade Scale

Category	Boulders, Cobbles	Gravel	Sand	Silt	Clay
Size range (mm)	64 – 4,000	2 – 64	0.062 – 2	0.004 – 0.062	0.00024 – 0.004

Source: Julien (1995).

Sediment size is also loosely discussed in terms of "fine sediment" and "coarse sediment." Fine sediment consists primarily of particles smaller than 0.85 mm. Coarse sediment consists primarily of particles from 0.85 to 9.5 mm (USEPA 2006b). Particles smaller than 1 micron (0.001 millimeter) in diameter are sometimes referred to as colloids.

Sediment samples rarely contain particles of a single size. Particle size distributions describe the percentage by weight of sediment materials in each size category for a given sediment sample.

Particle size helps to determine how quickly a particle will settle out of suspension in the water column onto the bed of a waterbody. The *fall* or *settling velocity* of sediment particles can be estimated using a form of Stokes' law (Julien 1995; Chapra 1997):

$$\omega = \alpha\left(\frac{g}{18}\right)\cdot\left(\frac{\rho_S - \rho_W}{\mu_m}\right)\cdot d_s^{\,2} \qquad \text{(Eq. 2-1)}$$

Where:

ω = settling velocity ($cm \bullet s^{-1}$)

α = dimensionless form factor accounting for geometry (α=1 for sphere)

g = gravitational acceleration (g=981 $cm \bullet s^{-2}$)

ρ_s = density of sediment particle ($g \bullet cm^{-3}$)

ρ_w = density of water ($g \bullet cm^{-3}$)

μ_m = dynamic viscosity ($g \bullet cm^{-1} \bullet s^{-1}$)

d_s = particle diameter (cm)

This formula indicates that settling velocity increases as particle size increases, assuming all other factors are equal. Stokes-type settling is most directly applicable to the behavior of sediment in detention basins and some reservoirs and lakes, in which flow velocities are low and turbulent energy in the water column is minimal. In streams and rivers, settling velocity operates in conjunction with turbulent energy affecting particle suspension to determine sediment settling dynamics. Since turbulent energy is required to offset the force of gravity (settling) in order to keep a particle in suspension, a greater amount of turbulent energy is required to suspend larger, heavier particles than for smaller particles.

The degree to which a particle remains suspended helps to determine the degree to which it will be able to move with water flow within a surface water or whether it will be deposited on a surface water bed. It also determines whether or not a sediment particle will contribute to surface water turbidity, since only suspended particles increase turbidity levels. Particles smaller than 0.063 mm (silt and clay) tend to remain suspended in flowing freshwater systems and are the primary contributors to water turbidity (USEPA 2006b). However, small sediments can also be transported in the aquatic environment in the form of aggregates bound together by living (e.g., bacteria and algae) and nonliving (e.g., organic detritus, extra-cellular polymeric substances) material (Bilotta and Brazier 2008). These aggregates have transport behaviors closer to particles coarser than the particles composing the aggregate. Coarser particles are more likely to settle from the water column, with the coarsest particles settling closest to their discharge point to surface water.

The sections below provide additional information on transport dynamics associated with several surface water types.

2.2.2 Sediment and Turbidity Behavior in Specific Waterbody Types

2.2.2.1 Sediment and Turbidity in Streams and Rivers

A distinguishing characteristic of streams and rivers is their strong, primarily unidirectional flow. Their transport of sediment is predominantly controlled by stream transport capacity and sediment physiochemical characteristics and supply rate. Larger sediments generally experience more episodic movement over longer time scales through watersheds. Smaller sediments generally move more continuously and within a shorter time scale. This difference is due to the fact that larger sediments rely on larger, more powerful flows for transport, which occur episodically and less frequently than flows able to move smaller particles.

Sediments transported by river and stream channels are typically described as bed load and suspended load. The boundary separating these categories changes with flow conditions. Sediments not in transport at a given time are often referred to as bedded sediments. If flow conditions change, these sediments can become bed load or suspended load. Sediment particle size is the primary distinguishing factor among these categories, with the largest particles deposited as bedded sediments, larger particles moving as bed load, and finer particles moving as suspended load. Total sediment load consists of the sum of bed load and suspended load.

Bed load consists of the movement of particles along the streambed by rolling, sliding, and a hopping process known as saltation which involves the brief suspension of sediment particles by turbulent flow. Particles moving as bed load often consist of sand and coarser size fractions. Bed load movement is intermittent in most settings, and bed load particles may not move for long periods between high flow events.

Suspended sediment load is the movement of sediment particles supported by turbulent motion in stream or river flow and often consists primarily of clay and silt size particles. Suspended sediment (and the turbidity it creates) move more quickly downstream with streamflow than bed load from a given discharge point. If the suspended sediment input to the stream or river is a single pulse, rather than continuous, the stream or river can often move it downstream within a relatively short time period.

Several hydraulic and geomorphologic factors determine stream transport capacity including channel width, flow depth and cross-sectional geometry, bed slope and roughness, and discharge velocity and volume. In general, the more turbulent energy available for suspension and mobilization of sediment, the greater the sediment transport capacity per unit of stream width and the larger the size of sediment particles that can be moved. Several empirical formulas have been developed to estimate sediment transport capacity on the basis of flow- and channel- related variables. One example, developed by Simons et al. (1981), takes the form of a power law:

$$q_s = c_1 \cdot h^{c_2} \cdot V^{c_3} \qquad \text{(Eq. 2-2)}$$

Where:

q_s = sediment transport per unit stream area (ft^2/s)

h = depth of flow in ft

V = flow velocity in ft/s

c_1, c_2, c_3 = empirical coefficients calibrated to reflect channel slope (gradient) and median particle size diameter.

The unit transport capacity is strongly related to flow velocity *(V)*. Higher velocity flows are able to move more sediment. Channels with lower stream gradients have, for a given volume of stream flow, lower stream water velocities which allow more time for settlement of sediment in that channel. Many, if not most, of the hydraulic factors controlling sediment transport capacity, including channel geometry, depth, slope, and velocity of flow, vary in time and space within a given river system.

Many North American rivers and streams possess a strong seasonal discharge cycle with spring discharge volumes typically many times larger than those of late summer and autumn flows. Intense or prolonged rainfall events can also generate flood pulses of hourly to daily duration, which often have significant turbulent energy. In addition, as Equation 2-2 indicates, sediment transport capacity is a nonlinear function of flow-related variables, so large flows have significantly greater transport energy. The movement of sediments, both as bed load and as suspended load, is thus highly nonuniform in time for most river systems. The majority of annual sediment flux, particularly the movement of coarse or highly cohesive sediment particles, may occur over a relatively short period of time during a single flood event. Between such events, sediments are typically stored within the stream or river channel. Episodic movement and deposition in stream channels between periods of movement has been documented for sediments in watersheds whose total sediment load has been substantially increased by construction activity. Sediment inputs from construction sites can outpace the immediate transport capacity of the stream and may not migrate downstream until a major flow event occurs (Lee et al. 2009).

Erosion and deposition of sediments within a river course also exhibit spatial patterns strongly related to stream morphology. River reaches with smaller cross-sectional flow area, steeper slopes, and higher flow velocities discourage the deposition of sediments. These traits tend to be characteristic of smaller streams and rivers in upper elevation catchments, often at the headwaters of larger watersheds. These higher-gradient streams may decrease their construction-associated sedimentation levels more quickly than other aquatic ecosystems due to the occurrence of high flows, particularly in spring or after large storms, that can resuspend and transport finer sediments downstream. Assuming cessation of additional sediment input upstream, these stream dynamics may eventually restore a naturally coarse-grained channel bed (Barton 1977; Berry et al. 2003), though some sediment may continue to persist in low-energy areas of the stream (e.g., shallow side pools). By contrast, wider channels with lower bed slopes and flow velocities act as regions of relative sediment deposition. Channel bottoms may be covered with finer sediments, in contrast to the exposed rocks, boulders, and gravels seen in the channel beds of higher-energy streams and rivers. Natural sediment deposition is more characteristic of channels at lower elevations in a watershed.

Stream and river hydraulic and geomorphologic variables provide one set of controls on sediment transport capacity. Sediment transport is also be regulated by the rate and quality of sediment supply (Julien 1995). Sediment supply can outpace, match, or fall below the ability of a channel to transport it. Within a particular reach, sediment fluxes can originate from land surface erosion, streambank erosion, upstream reach sediment input, or remobilization of sediments previously deposited within the reach. Channels whose sediment supplies outpace their transport capacity will accumulate sediments. The size of a channel can decrease as sediments accumulate, increasing the likelihood of flooding and other overbank flow events. Channels with sediment supplies falling below transport capacity will work to mobilize additional material from channel beds and banks.

In all streams, sediments are preferentially deposited in regions of low-energy flow, including pools and the inside of bends (Chapra 1997). If sufficient quantities of sediment are deposited, the deposition features can alter channel morphology and flow patterns, obstruct flow, and exacerbate flood events. Increased sediment supply during construction activity has converted some naturally meandering streams to braided or straighter, more channelized forms (Paul and Meyer 2001). Fine sediments deposited on stream and river beds may also impede water exchange with groundwater sources (both recharge and discharge) (USFWS 1998). Sediment deposits can also provide substrate for the growth of plants in channels in locations where they would normally not occur. King and Ball (1964) and Taylor and Roff (1986) documented this effect downstream of highway construction sites.

Individual sediment deposits are often not permanent features of streams and rivers, since they can be scoured and moved downstream during major flow events. Streams and rivers can also flow outside their normal channels during major flow events and deposit sediments on low-lying areas adjacent to the channel such as banks, floodplains, and terraces. These sediments may, at a later time, be remobilized during an even larger flow event.

Lee et al. (2009) documented that watersheds undergoing construction activity may take many years or even decades to move sediments discharged from construction sites fully through watersheds. Sediment yields from larger watersheds may therefore remain elevated for some time after the implementation of enhanced sediment and erosion control measures and after the completion of most construction in the watershed.

Within the time scale of a single precipitation event, a certain amount of time is also necessary to move suspended sediment and turbidity from its original point of entry into the surface water network to downstream waters. If construction is occurring primarily in the headwaters of a watershed, turbidity and suspended sediment levels immediately downstream of construction sites will often decline within relatively short periods of time following the cessation of precipitation events (assuming discharge from the site is primarily precipitation-driven and not significantly attenuated). Turbidity and suspended sediment levels further downstream, however, can remain elevated even after flow levels return to normal because of the time required for their transport downstream.

An additional dynamic influencing short-term observation of impacts from upstream construction activity is that levels of turbidity and sediment deriving from construction activity tend to decrease as they move downstream from a construction site through the surface water network. This decrease is due to sediment deposition during transport and dilution by waters containing lower levels of sediment and turbidity than construction site discharges (Wolman and Schick 1967; Lee et al. 2009).

2.2.2.2 Sediment and Turbidity in Lakes, Reservoirs, and Ponds

A number of lakes, ponds, and reservoirs have a unidirectional flow component and therefore have some of the basic dynamics described for rivers and streams. However, most lakes, ponds, and reservoirs have lower flow velocities, greater depth of flow, and longer water residence times than streams and rivers and therefore act as deposition zones (sinks) for sediments. Longer flow residence times also mean that influxes of turbid water can linger for longer periods of time than they would in a stream or river. Residence times vary widely among lakes, ponds, and reservoirs.

The lake and reservoir flow environment more closely approximates the still-water conditions under which Stokes' law (*Section 2.2.1*) applies to sediment particles to predict settling velocity. The assumption of Stokes-type settling is often used to predict the rate of sedimentation in lakes and reservoirs by comparing the time required for a particle to settle with the particle's transit time through the waterbody. Several surface water characteristics influence settling and transit times, including size, shape, depth, and regulation of outflow (Chapra, 1997). These characteristics vary widely among lakes, ponds, and reservoirs.

Particle transit time also varies with flow. Higher flows generally create shorter transit times as higher volumes of water move more quickly through a surface water. Higher flows often transport both coarser particles, which are more likely to settle in a lake, pond, or reservoir, and large quantities of fine sediments, which may not have sufficient time to settle during a higher flow period. Sediments unable to settle in lakes, ponds, and reservoirs due to insufficient settling time or turbulence from water flow, wind, or human activity can continue their transport downstream.

Given sufficient residence time for incoming flows, however, these waters can remove significant quantities of sediment and turbidity from incoming waters (Barton 1977). The deposition process decreases sediment and turbidity loads to streams and rivers and other surface waters downstream (Renwick et al. 2005; Lee et al. 2009), though it increases sedimentation in the lake, reservoir, or pond. Over time, sediment accumulation reduces a waterbody's volume and decreases its capacity for sediment removal.

Sediment deposition in lakes and reservoirs often begins with the formation of deltas at the point of water inflow, where incoming stream flow decelerates and the heaviest particles settle (Julien 1995). Turbulence from wind, currents, and human activity can keep finer sediments in suspension, creating a deposition pattern called focusing in which coarser particles are deposited in shallower waters and finer particles in deeper waters, often in the centers of lakes (Chapra 1997). Given sufficient sediment deposition, ponds and shallow areas in lakes and reservoirs may be transformed into wetland environments.

2.2.2.3 Sediment and Turbidity in Wetlands

Wetlands are natural sediment deposition zones. Water flow volumes and velocities are smaller than in most other surface water types, allowing time for settling of sediment from the water column. Because of lower flow-through volumes and speeds, influxes of turbid and sediment-laden water typically remain for a longer period of time in a wetland than they do in a stream or river channel.

Wetland ecosystem structure is heavily influenced by the type of vegetation a wetland supports, which in turn is influenced by water and sediment distribution in the wetland. Elevated sedimentation levels can change the type of vegetation able to persist in a given wetland, with consequences for the organisms it no longer supports. In severe sedimentation cases, excess sediment may partially or completely fill in a wetland, creating dry land conditions.

2.2.2.4 Sediment and Turbidity in Estuaries

Estuaries are like lakes and reservoirs in that they vary widely in benthic geometries, residence times, flushing rates, vertical mixing, stratification, wave exposure, and other factors that govern sediment transport and deposition. Estuarine flows transition first from unidirectional turbulent channel flow controlled primarily by topographic gradient and discharge rate to a tidal river reach zone in which downstream flow is influenced by the tidal cycle. The flow regime transitions again in the estuary proper, in which water discharge and tidal forces offset each other. Flow regime changes again in the bay or open ocean where tidal, wave, or a combination of these forces dominate.

As turbidity and sediment-laden freshwater decelerates and encounters tidal cycles in the estuary, a null zone is formed in which channel discharge and tidal action largely cancel each other, favoring the deposition of suspended sediment (Chapra 1997). In addition, increasing salinity levels favor flocculation of fine sediments. Large deltas may form where rivers deposit sediments in coastal waters. Sediment deposited in estuaries can be disturbed and redistributed due to natural events (e.g., floods, high winds, tidal action) or by human activity (e.g., dredging).

2.3 Aquatic Life Impacts of Sediment and Turbidity

Numerous aquatic ecosystem impacts from construction site discharges have been documented (see *Chapter 4*).This section summarizes studies of aquatic organism and ecosystem impacts associated with elevated sediment and turbidity levels in surface waters. EPA has reviewed available literature on the biological effects of suspended and bedded sediments (SABS) and turbidity. This review is not an exhaustive summary of the available literature, which is extensive, but instead provides an overview of more commonly noted impacts.

A variety of organisms, including aquatic plants, invertebrates, amphibians, and fish, are affected by elevated sediment and turbidity levels. High levels of sediment and turbidity affect aquatic ecosystems by reducing photosynthetic activity, reducing food availability, burying habitat, and directly harming organisms. Organisms may relocate, sicken, or die. Organism loss can alter the composition of the aquatic community.

Both the magnitude and duration of sediment and turbidity exposure are important. Organisms vary in their ability to tolerate and recover from exposure. The proportion of organisms able to tolerate (or escape) periods of elevated sediment and turbidity levels can increase in impacted surface waters, while the proportion of sensitive species declines.

The appropriate level of sediment or turbidity varies from waterbody to waterbody. Some aquatic ecosystems contain organisms adapted to higher levels of sediment or turbidity (e.g., some large coastal floodplain rivers), whereas other ecosystems contain organisms adapted to lower levels of sediment or turbidity (e.g., alpine lakes, headwater streams). Smaller streams tend to have naturally clearer water, particularly those in forested watersheds. Short-term increases in suspended sediment and turbidity levels can naturally occur during spring thaws, storms, and other high flow events. However, even organisms adapted to sediment or turbidity influx (whether episodic or constant) can be harmed if input levels rise excessively and/or if their resiliency is taxed by other stressors.

A construction project can change natural sediment and turbidity dynamics by elevating sediment and turbidity levels significantly beyond those associated with natural events, for longer periods of time, and at times when an aquatic ecosystem and its organisms are unaccustomed to receiving such inflows (e.g., late summer low flow periods). Some waterbody types are better able to flush excess suspended sediment, turbidity, and deposited sediment (e.g., high energy streams), whereas others may retain accumulated sediments for years (e.g., lakes and wetlands) (see *Section 2.2*).

Sediment impacts have been researched through both laboratory dose-response and in-stream field studies. Much research has been done on the effects of sediment on fish (salmonids, in particular, because of their economic importance and sensitivity). More studies have been conducted on suspended sediment versus bedded sediment effects on organisms. Stream and coral reef habitats have been studied more intensively than river, lake, and estuarine habitat, though available data indicate that biota sensitive to elevated sediment levels exist in all of these environments. Many habitats with moderate and variable levels of sediment also need additional research (Berry et al. 2003).

Though not addressed in detail in this discussion, sediments and turbidity can also affect chemical, fungal, and microbial decomposition processes in aquatic ecosystems. Depending on the source and nature of the sediments, sedimentation can change the organic content of surface water sediment, which can alter fungal and microbial community activity. Rapid sediment deposition over an organic sediment layer can also cause a shift to anaerobic conditions in the isolated strata and an accompanying shift in microbial metabolism, methane generation, and nutrient dynamics (Tornblom and Bostrom 1995). In wetlands, depressed levels of algal and microbial biomass and density can indicate chronically elevated sedimentation levels (USEPA 1995).

The studies cited in this section attribute observed impacts to sediment and turbidity discharges in general, rather than to the specific source materials from which the sediments and turbidity derived. The discussion in this section provides a qualitative summary of the types of organism impacts associated with elevated sediment and turbidity levels. Berry et al. (2003) discuss the possibilities, given available data, for determining quantitative relationships (e.g., dose-response models) between sediment levels and organism responses. For most organisms, there is insufficient information currently available to create complete dose-response models (Berry et al. 2003). A model for clear water fish is discussed in *Section 2.3.3*.

The examples below discuss effects of elevated sediment and turbidity levels acting on organisms in isolation. However, many construction sites have been documented to discharge other pollutants in addition to sediment and turbidity such as nitrogen, phosphorus, metals, and other organic and inorganic compounds (see *Chapter 3*). A number of these other pollutants travel with sediment as it erodes and moves through the surface water network. In addition, many construction sites discharge to surface waters already impacted by stressors including pollutant discharges from other sources, invasive species, and flow modification. Interactions among sediment, turbidity, and other stressors and their cumulative effects on aquatic organisms have not been fully characterized (Berry et al. 2003).

Documented effects and relevant studies are organized in the discussion below by the type of aquatic organism studied (primary producer, invertebrate, fish, and other wildlife) and by the type of organism reaction. A section specific to threatened and endangered species follows this discussion (*Section 2.3.5*).

2.3.1 Primary Producers

Primary producers, including algae, phytoplankton, submerged aquatic vegetation, and other macrophytes (plants with large leaves), are found in most aquatic ecosystems. Aquatic ecosystems vary in the degree to which they derive energy and resources from primary producers. Some systems (e.g., small streams in forested watersheds) derive a large proportion of their energy from external sources (e.g., leaf debris) (Bilotta and Brazier 2008). Other systems are heavily dependent on the productivity of primary producers (e.g., sea grass beds).

Primary producers can grow in a variety of forms: rooted to surface water substrate, free-floating, or attached to rocks, aquatic plants, or other structures. The term periphyton refers to the algae, small plants, bacteria, microbes, and detritus attached to submerged surfaces in aquatic ecosystems.

Primary producers transform sunlight into energy and provide food for other aquatic organisms. Aquatic macrophytes and algae influence water column chemistry, including dissolved oxygen, pH, and nutrient levels; provide important habitat for many organisms; and reduce waterbody flow velocity. Loss of algae and macrophytes due to elevated sediment and turbidity levels can compound these pollutants' effects because fewer plant structures are available to slow water flow and facilitate settling of suspended sediment from the water column. This dynamic can allow elevated suspended sediment and turbidity levels to persist further downstream (Wood and Armitage 1997).

Turbidity is an important parameter for aquatic ecosystems because of its influence on the compensation point (the depth at which carbon production from photosynthesis equals consumption through respiration in aquatic plants). A decrease in a surface water's compensation point can translate into a decline in an aquatic ecosystem's primary producer community or a shift in its species composition. These changes can affect the ecosystem's overall species composition, productivity, and health.

Impacts to primary producers have been noted in some studies of construction site impacts to surface waters. Construction sites documented in these studies were discharging elevated levels of sediment and turbidity. King and Ball (1964) documented a decline in primary productivity levels. Cline et al. (1982) noted an increase in periphyton sediment content and a decline in periphyton abundance and algal diversity.

Several ways in which turbidity and sediment affect primary producers are described below.

2.3.1.1 Light Reduction

The growth and distribution of macrophytes, algae, phytoplankton, and coral zooxanthellae can be limited by excessive turbidity (Berry et al. 2003). Macrophytes with leaves that emerge above the water's surface are much less affected by reduced light penetration than submerged species. Reduced primary production due to light limitation from elevated turbidity is a common impact that has been documented by LaPerriere et al. (1983), Rivier and Seguier (1985), and Lloyd (1987) (all as cited in Waters 1995), Meyer and Heritage (1941) and Edwards (1969) (both as cited in Kerr 1995).

LaPerriere et al. (1983) found that moderate turbidity levels of approximately 170 NTU reduced oxygen production in a stream by 0.08–0.64 g/m^2 per day relative to clear water conditions. Above 1,000 NTU, primary production was undetectable. Lloyd (1987) developed a model relating turbidity and primary production. The model indicated that in a clear, shallow stream, a turbidity increase of 5 NTU decreased primary production 3 to 13 percent. A 25 NTU increase reduced primary production up to 50 percent. The study documented stronger effects from turbidity in lakes due to greater water depths and the greater importance of the phytoplankton community. Quinn et al. (1992, as cited in Bilotta and Brazier 2008) found a 40 percent reduction in phytoplankton biomass exposed to suspended sediment concentrations of 10 mg/L for 56 days.

Guenther and Bozelli (2004) investigated whether lower light levels or increased sinking of phytoplankton due to binding to sediment particles was the cause of their reduced photosynthetic activity in turbid waters. The authors identified lower light levels as the primary cause.

Stony corals typically live in clear, oligotrophic waters where their symbiotic association with photosynthetic zooxanthellae provides more than 100 percent of the coral's daily metabolic carbon requirements (Muscatine et al. 1981; Grottoli et al. 2006). Corals exposed to elevated turbidity levels have reduced photosynthesis rates and must compensate for this loss of resources (Philipp and Fabricius 2003).

Sediments can also reduce photosynthetic activity by settling on and coating macrophytes and periphyton. Even a thin layer of sediment can block enough light to inhibit the growth of algae attached to surface water substrates (Welsh and Ollivier 1998). Lower primary production by macrophytes and periphyton can result in their decline and replacement with suspended algal communities. A shift in primary producer community composition can alter a surface water's invertebrate and fish community composition.

2.3.1.2 Direct Physical Damage

Algae and aquatic macrophytes can suffer physical damage from elevated suspended sediment levels (Waters 1995). Lewis (1973, as cited in Kerr 1995) conducted an experiment in which suspended coal particle concentrations greater than 100 mg/L caused severe abrasive damage to the leaves of the aquatic moss *Eurhynchium riparioides* over a period of 3 weeks. The damage substantially lowered the moss's ability to produce chlorophyll and photosynthesize. Periphyton is also susceptible to being scoured from stream substrate by suspended sediment particles (Welsh and Ollivier 1998). Francouer and Biggs (2006) investigated the interaction of flow velocity and suspended solids in benthic algal communities and found that high suspended sediment concentrations further increased algae removal above that due to flow alone. Some taxa were more susceptible to removal than others. The results indicated that suspended sediment scour may be an important mechanism for algae removal during flood events and that some variability in biomass removal among flood events and benthic algal composition may be the result of differences in suspended sediment load. Birkett et al. (2007, as cited in Bilotta and Brazier 2008) found that suspended sediment levels of 100 mg/L stimulated periphyton growth and filament length under low flow conditions, but levels of 200 mg/L significantly reduced biomass and filament length.

Macrophytes, algae, and periphyton can also be buried and damaged or killed by excessive sedimentation, with different species having varying abilities to cope (Berry et al. 2003). Some organisms can outgrow sediment damage, though this growth may consume additional organism resources. A reduction or complete elimination of periphytic material and a reduction in component diversity due to elevated sediment levels is described by Van Nieuwenhuyse and LaPerriere (1986), Pain (1987) (both as cited in Waters (1995), and Samsel (1973, as cited in Kerr 1995).

2.3.1.3 Substrate Modification

Sediment deposition in surface waters can change the types of algae (Wood and Armitage 1997) and macrophytes able to grow in a waterbody by modifying nutrient availability and substrate chemistry, texture, stability, and, depth. In many waterbodies, the locations of macrophyte beds are highly correlated with the spatial distribution of fine-grained sediments since these often provide better rooting substrate. Sediment deltas that form rapidly as a result of sediment runoff to a waterbody are often susceptible to colonization by nuisance invasive macrophyte species.

Some established plants respond to elevated sedimentation levels by extending rhizome growth upwards, but such growth patterns are abnormal for some species and consume plant energy (USFWS 1998). In some cases, plants may not be able to grow quickly enough to adjust to increased sedimentation. Over time, substrate may change to the extent that it no longer supports the original primary producer community. Wetlands are particularly vulnerable to changes in sedimentation rate.

2.3.2 Invertebrates

Aquatic invertebrates, including zooplankton, insects, crustaceans, and bivalves (e.g., mussels and clams), are widely distributed among aquatic ecosystems and can be found in even the smallest surface waters. They provide an important link in the aquatic food chain between primary producers and larger organisms such as amphibians, reptiles, fish, birds, and mammals (Waters 1995).

Because many benthic macroinvertebrates are relatively stationary and have limited powers to evade polluted waters, their abundance, diversity, and species composition is widely used as an indicator for overall aquatic ecosystem health (USEPA 2006d). The effects of elevated sediment and turbidity levels on invertebrates, particularly on benthic macroinvertebrates, have been the focus of a large number of studies over the past several decades.

Elevated sediment and turbidity levels impact invertebrates by reducing their health, disease and parasitism resistance, and abundance, as documented in literature reviews conducted by Kerr (1995), Waters (1995), and Henley et al. (2000). The density, diversity, and structure of aquatic invertebrate communities can change as a consequence, to contain a higher proportion of sediment-tolerant species. Gammon (1970, as cited in Kerr 1995) finds that TSS concentrations between 40 and 120 mg/L reduced macroinvertebrate density by 25 percent, and greater TSS concentrations reduced density by 60 percent. Organisms intolerant of turbid waters include mayflies and other Ephemeroptera, stoneflies (Plecoptera), caddisflies (Trichoptera), clam, and bryozoan species (Cooper 1987, as cited in Kerr 1995).

The severity of sediment and turbidity impacts on invertebrate populations is related to the intensity and duration of exposure (Anderson et al. 1996, as cited in USEPA 2004a). Some populations are adapted to the short-term increases in suspended sediment and turbidity levels that can occur during spring thaws, storms, and other natural high flow events. However, high or sustained elevated levels of sediment and turbidity can alter long-term community structure by degrading habitat, food sources, organism health, and increasing “drift” (i.e., voluntary/involuntary release of substrate by aquatic macroinvertebrates with subsequent transport downstream)

If disturbance to a macroinvertebrate community is intense enough, it may impact the ability of the community to recover its former condition, particularly if sensitive organisms are eliminated from surrounding areas, as well, or habitat has been modified to the point where it is no longer able to support the former community (Montgomery County DEP 2009). Healthy invertebrate populations in unaffected parts of a surface water system can provide organisms to recolonize disturbed areas once construction activity has ceased.

Multiple studies have documented impacts to invertebrates from construction site sediment and turbidity discharges (King and Ball 1964; Peterson and Nyquist 1972; Barton 1977; Reed 1977; Chisholm and Downs 1978; Extence 1978; Lenat et al. 1981; Tsui and McCart 1981; Cline et al. 1982; Taylor and Roff 1986; Young and Mackie 1991; Ohio EPA 1996a, 1997f, 1998d, 1999d, 2003a, 2006b; Reid and Anderson 1999; Fossati et al. 2001; Levine et al. 2003, 2005; Hedrick et al. 2007; Chen et al. 2009; Montgomery County DEP 2009).

Several categories of impact identified by Kerr (1995) and Wood and Armitage (1997) are described below.

2.3.2.1 Loss of Habitat

Increased sediment and turbidity levels in waterbodies can degrade aquatic invertebrate habitat, including pools and interstitial spaces in waterbody substrate and wetland areas. Habitat alteration due to changes in substrate composition can modify the distribution and density of different invertebrate species and therefore the structure of an invertebrate community (Waters 1995; Berry et al. 2003). Sediment settles from the water column onto the substrate to which sedentary organisms attach and also fills small crevices in waterbody beds (interstitial spaces) where a variety of species shelter, reproduce, feed, and seek protection from predators, high flow velocities, and low flow conditions. Large accumulations of sediment are also anoxic at depth. When interstitial spaces fill with sediment, the microhabitat diversity of the substrate significantly declines, as does the diversity of organisms it can support (USEPA 2006d).

Embeddedness refers to the extent to which rocks or organic debris in a surface water are covered or sunken into fine bottom sediments and is an important physical component of stream or river habitat (USEPA 1999). Increased embeddedness due to sedimentation indicates habitat degradation. Studies have found high correlations between benthic invertebrate response to depth and degree of embeddedness (Berry et al. 2003). Correlations have also been found between benthic invertebrate abundance and substrate particle size (Waters 1995). Even small increases in sedimentation levels can impact caddisfly pupa survival (Berry et al. 2003). Ryan (1991, as cited in Henley et al. 2000), concludes that as little as a 12 to 17 percent increase in interstitial fine sediment could result in a 16 to 40 percent decrease in macroinvertebrate abundance.

Siltation of previously coarse substrates is also implicated in the reduction of mussel populations by Ellis (1936) and Marking and Bills (1980) (both as cited in Waters 1995). Very thin veneers of sediment can decrease the settlement and recruitment of some bivalve larvae (Berry et al. 2003). One study postulates that many of the more than 30 extinct species of North American mussels became extinct due to a loss of habitat, most likely from sedimentation of surface water substrate (Henley et al. 2000). In coral reef systems, most coral larvae seek firm substrates and will not settle and establish themselves on shifting sediments (Berry et al. 2003).

Invertebrates can also be indirectly affected by impacts to primary producers in an aquatic community. A shift in primary producers can impact the structure of invertebrate community by changing food and shelter availability and quality. As discussed in the previous section, increased turbidity and sediment levels can adversely affect the primary producer community and change its abundance and composition. As turbidity increases, the contribution of primary production from aquatic macrophytes may decline, while that associated with phytoplankton comes to dominate the system. Sediments and turbidity can adversely affect the growth, abundance, and species composition of the periphytic algal community, reducing food availability for invertebrate grazers that feed predominantly on periphyton. A shift in primary producers can impact the structure of the invertebrate community by changing food and shelter availability and quality. This can lead to shifts in the relative proportion of various feeding guilds and habitat preferences in the resulting aquatic invertebrate community (Vannote et al. 1980; Merritt and Cummins 1996).

2.3.2.2 Altered Movement and Increased Drifting

Movement upstream or downstream ("drift") of benthic macroinvertebrates has been found to increase with elevated suspended sediment and turbidity concentrations (Kerr 1995; Berry et al. 2003). Increases in invertebrate drift have been shown to be specifically related to the turbidity caused by suspended sediments (Waters 1995). One study found that suspended sediment levels of 120 mg/L were sufficient to cause drift (Berry et al. 2003). Rosenberg and Wiens (1978, as cited in Bilotta and Brazier 2008) found increased benthic macroinvertebrate drift after 2.5 hours of exposure to 8 mg/L suspended sediment. Increased drift leads to increased predation on the displaced benthic invertebrates (Waters 1995) and causes a decline in invertebrate abundance in the impacted surface water (White and Gammon 1977, as cited in Waters 1995). Benthic macroinvertebrate drifting downstream of construction sites for pipeline stream crossings has been documented in several studies (Tsui and McCart 1981; Reid and Anderson 1999).

2.3.2.3 Physiological Changes

The physical effects of elevated sediment levels on invertebrate species can be severe. Sediment particles can clog an organism's filter feeding apparatus, filtration system, respiratory system, and digestive system (Kerr 1995), resulting in compromised health or death of the organism. Affected organisms include taxa with fragile or rudimentary gill structures, such as stoneflies (Merritt and Cummins 1996) and organisms that primarily graze in the water column (e.g., zooplankton). Alabaster and Lloyd (1982, as cited in Bilotta and Brazier 2008) found that Cladocera and Copepoda gills and guts were clogged after 72 hours of exposure to suspended sediment levels of 300 to 500 mg/L.

Most species of coral are highly intolerant of sedimentation and will expend energy to flush excess sediments. If a coral is unable to compensate for excess sedimentation, tissues can smother and die. Sediments can also reduce zooxanthellae photosynthesis, reduce growth rates, cause temporary bleaching, and eventually death. These impacts can affect other parts of the coral reef food web. Consequently, increasing sedimentation levels are associated with reductions in coral reef diversity (Rogers 1990), alterations in community composition by growth morphology (Rogers 1990, Stafford-Smith and Ormond 1992), decreased skeletal extension rates (Dodge et al. 1974, Miller and Cruise 1995, Heiss 1996), and reduced coral recruitment (Rogers 1990, McClanahan et al. 2002, Fabricius et al. 2003). Other coral reef organisms, such as sponges, are thought to be sensitive to sedimentation as well (Berry et al. 2003).

2.3.2.4 Impaired Feeding Activity

Another effect of elevated suspended sediment and turbidity levels on aquatic invertebrates is a reduction in their feeding activity or the efficiency of their feeding. Kerr (1995) noted that this impact is predominant in filter feeding species such as mussels and clams and cited Ellis (1936), who found that under turbid conditions, mussels and clams reject silt-laden food and, under highly turbid conditions, close their shells completely. Waters (1995) also discussed sediment's impact on feeding, citing Ellis (1936) and Aldridge et al. (1987), who found that sediment additions impaired feeding in three species of bivalves. Reactions to elevated sediment levels by freshwater mussels may be species specific. Mussels compensate for increased suspended sediment levels by increasing filtration rates, increasing the proportion of filtered material that is rejected, and increasing the selection efficiency for organic matter. This process increases the metabolic cost of feeding per unit of food and decreases resources available for growth and reproduction.

Species that have evolved in turbid environments may be better able to select between organic and inorganic particles during feeding. For example, one study found that a mussel from rocky coastal environments (*Mytilus californianus*) was more sensitive to elevated suspended sediment levels than a mussel of the same genus from a bay environment (*Mytilus edulis*), which is typically more turbid (Berry et al., 2003). Many endangered freshwater mussel species evolved in fast flowing streams with historically low levels of suspended sediment. Compared to other species that evolved in more turbid environments, such rocky intertidal habitat species may not be able to differentiate between organic and inorganic particles as well.

Light attenuation due to elevated turbidity can reduce feeding efficiency of invertebrates. Copepods and daphnids have been found to reduce feeding activity in response to elevated suspended sediment levels (Berry et al. 2003). Sediment coatings on macrophytes and periphyton can reduce their quality as food sources because herbivores must consume elevated levels of sediment with the plant material (Ryan 1991).

Kerr (1995) cited three studies (Paffenhofer 1972; Appleby and Scarratt 1989; Kirk 1992) that found reduced growth rates in invertebrates subjected to elevated sediment levels due, potentially, to reduced feeding activity and/or reduced food value. Graham (1990, as cited in Waters 1995) finds that a decrease in the percentage of organic matter in periphytic material may result from additional sediment particles binding to the periphyton.

2.3.2.5 Direct Mortality

Sediment can directly or indirectly increase invertebrate mortality levels. Forbes et al. (1981, as cited in Kerr 1995) found that elevated suspended sediment levels increased amphipod (shrimp-like crustacean) mortality. Henley et al. (2000) concluded from a literature review that increased sediment concentrations have a negative effect on the survival rates of freshwater mussels with the magnitude of the impact being species specific. Robertson (1957, as cited in Bilotta and Brazier 2008) found the survival and reproduction of Cladocera to be harmed by 72 hours of exposure to suspended sediment concentrations of 82 to 392 mg/L.

Organisms can also be buried and smothered by heavy sedimentation. Large sediment accumulations become anoxic at depth. Species vary in their ability to evade burial under moderate sedimentation deposits. Sediment grain size, depth, and bulk density and species motility, living position, and tolerance of anoxic conditions during burial affect this ability (Berry et al. 2003).

Sediment organic matter is an important food source for many benthic macroinvertebrates. A number of toxic pollutants bind to and travel with sediment and can be ingested by invertebrates with the organic matter (Paul and Meyer 2001). The aquatic organism effects associated with these toxic pollutants are discussed in *Chapter 3*.

2.3.3 Fish

A substantial amount of research has been conducted to identify and quantify the effects of elevated sediment and turbidity levels on a variety of fish species, more so than other taxonomic groups (Berry et al. 2003). Salmonids, in particular, have been well studied because of their commercial and recreational importance and because of the concerns and well-documented impacts arising from logging in the Pacific Northwest and elsewhere (Waters 1995).

The level of impact from suspended sediments and turbidity is a function of the interaction of many factors, including natural sediment and turbidity levels for the area in question; pollutant concentration; exposure duration; particle size; ambient temperature; physical and chemical properties of the sediments; associated toxins; co-occurring stressors; and fish acclimatization, life history stage, migration, and breeding season (Anderson et al. 1996, as cited in USEPA 2004a). In general, longer duration and higher level exposures produce greater effects. Milder, primarily behavioral effects are observed at lower magnitude and duration exposure levels. As exposure and duration levels increase, effects become sublethal and lethal (Berry et al. 2003). Several researchers have sought to construct models linking varying levels and durations of sediment exposure to fish responses, but responses vary widely among different species, and data are generally insufficient at this time to fully characterize many of them (Berry et al. 2003).

Newcombe (2003) has created a model for clear water fishes that relates magnitude and duration of exposure to turbid water to a variety of adverse effects. The model indicates that fish exposure to turbidity levels of 7 to 1,100 NTU for 1 to 48 hours can create effects ranging from small behavioral impacts to death of a portion of the exposed population. This model does not describe fish that typically inhabit more turbid surface waters and that exhibit a different set of reactions to a given range of turbidity levels and exposures.

Short-term increases in suspended sediment and turbidity levels can naturally occur during spring thaws, storms, and other natural events. Aquatic ecosystems also vary in their natural sediment and turbidity levels. Studies have found that, in general, fish typically found in environments with naturally high turbidity preferred more turbid environments in the laboratory (Berry et al. 2003). Other fish species are very sensitive to elevated sediment levels (e.g., entire salmonid fisheries have been destroyed by elevated sediment levels) (Berry et al. 2003).

A construction project may produce sediment and turbidity levels significantly higher those associated with natural events, for longer periods of time, and at times when an aquatic ecosystem is unaccustomed to receiving such sediment inputs. Many fish species have seasonal migration and breeding behaviors which can be disrupted by elevated sediment levels. Younger fish are more vulnerable to sediment impacts and are more prevalent within a population at certain times of the year (Berry et al. 2003).

Observed impacts from elevated sediment and turbidity levels fall into several broad categories, discussed below. The potential cumulative effect of these impacts includes reduced disease and parasite resistance, reduced growth, and degraded health of individual organisms in the fish community. These impacts may decrease fish population levels in affected areas. Reductions can take place both through direct mortality in the short term and reduced reproductive success in the long term. Newcombe (1994) and Newcombe and Jensen (1996) (both as cited in Henley et al. 2000) found that elevated sediment concentrations are associated with increased mortality in at least 14 species of fish. EPA (USEPA 2004a) states that suspended sediment concentrations between 500 and 6,000 mg/L can result in significant (greater than 50 percent) mortality. Berkman and Rabeni (1987, as cited in Wood and Armitage 1997) noticed a decline in the overall abundance of fish stocks as sediment levels increased in a river in northeastern Missouri. The overall impact of these effects can be a change in fish community composition to one composed predominantly of species that are tolerant of sediment and turbidity, primarily bottom-feeders (Berry et al. 2003), or an overall reduction in fish abundance.

Impacts to fish communities downstream of construction sites discharging elevated levels of sediment and turbidity have been documented in multiple studies (King and Ball 1964, Graves and Burns 1970, Barton 1977, Garton 1977, Reed 1977, Werner 1983, Taylor and Roff 1986, Ohio EPA 1997f, Ohio EPA 1999c, Reid and Anderson 1999, Hunt and Grow 2001, California Department of Fish and Game 2004, Montgomery County DEP 2009).

2.3.3.1 Avoidance and Other Behavioral Responses

The severity of behavioral responses is associated with the timing of disturbance, the level of stress, fish energy reserves, phagocytes, metabolic depletion, seasonal variation, and habitat alteration (USEPA 2006b).

Because fish are generally more mobile than aquatic plants, plankton, and invertebrates, some species are capable of actively avoiding waters with high levels of suspended sediment or turbidity if migration routes to alternative habitats are available (mobility may be limited by culverts, small dams, and other obstacles). While larger, nonlarval fish are generally capable of avoidance behavior (Berry et al. 2003), smaller and younger fish are generally less mobile and therefore more vulnerable to elevated sediment and turbidity levels. Avoidance behaviors may reduce or eliminate fish populations in stream reaches with sustained elevated sediment levels. Barton (1977) noted a relocation of fish populations away from areas with elevated TSS levels due to construction near a small stream. Servizi and Martens (1987, as cited in Reid et al. 2003) determined that turbidity levels in excess of 37 NTU elicit avoidance behavior among fish populations. Berg (1982, as cited in Bash et al. 2001) documented that juvenile coho salmon exposed to a short-term pulse of 60 NTU water left the water column and congregated at the bottom of the test tank, returning to the water column once turbidity was reduced to 20 NTU. Boubee et al. (1997, as cited in Newcombe 2003) found inhibition of upstream migration of one species at turbidity levels of 15 to 20 NTU. Because of avoidance behaviors, some fish may be excluded from otherwise desirable habitat due to increased turbidity (Berry et al. 2003).

Other fish behavioral responses to elevated sediment and turbidity levels include increased frequency of the cough reflex and temporary disruption of territoriality (USEPA 2006b).

2.3.3.2 Feeding and Hunting

Elevated suspended sediment and turbidity levels have been shown to reduce feeding rates in several species of fish (Kerr 1995). Elevated turbidity reduces the prey reaction distance for trout and therefore reduces foraging success (Hedrick et al. 2006). This effect is generally believed to be a result of decreased visibility caused by turbidity because many fish use vision to locate food. De Robertis et al. (2003) found that turbidity level increases affected piscivorous fish feeding more than planktivorous fish feeding. Piscivorous fish visually locate food sources at much longer distances than planktivorous fish do. EPA (USEPA 2004a) concluded that turbidities greater than 25 NTU or TSS levels of 2,000–3,000 mg/L or greater are sufficient to reduce fish predation abilities. Sweka and Hartman (2001) observed that brook trout prey reaction distance decreased curvilinearly as turbidities levels increased from 0 to 43 NTU. Although most fish species' predation abilities are reduced by increased turbidity levels, some fish species have been documented to hunt better as turbidity increases because of increased contrast between the prey and surrounding water. This advantage deteriorates as turbidity levels further increase, however (Berry et al. 2003). Some larval fish species (e.g., striped bass - *Morone saxatilis*) appear to be able to feed under extremely turbid conditions (Berry et al. 2003).

Reduction in predation can benefit prey species able to use turbid water to hide from predators (Rowe et al. 2003). However, a paper by the Canadian DFO (2000) suggests that as turbidity levels continue to increase, the advantage of reduced predation is outweighed by the prey species' own increased difficulty in locating food.

An additional effect on feeding and hunting from elevated sediment and turbidity levels is decreased food availability and quality due to invertebrate community impacts, decreased primary productivity, and sediment coatings on macrophytes and periphyton, which increase the quantity of nonnutritive sediment herbivores must consume (Waters 1995; Ryan 1991). Bottom-feeding fish can be adversely impacted if sediment organic content and associated microbial and macroinvertebrate communities are degraded by sedimentation.

2.3.3.3 Breeding and Egg Survival

Elevated sedimentation levels can reduce spawning habitat for multiple fish species, particularly benthic spawners. Many fish species use well-aerated interstitial spaces in surface water beds to lay eggs. Sedimentation impact levels vary with differences in species sensitivity, life stage impacted, sediment particle size, and sedimentation rates and total magnitude. Eggs and larvae can be buried too deeply by sediment to survive (Berry et al. 2003). Sediments that settle onto surface water beds, particularly finer-sized particles, can reduce the level of dissolved oxygen available to eggs deposited in the substrate (Kerr 1995). Even thin layers of sediment only a few millimeters thick can disrupt the normal exchange of gases and metabolic wastes between eggs and the water column (Berry et al. 2003). Bjornn et al. (1977), as cited in Kerr (1995), found that successful emergence of fish fry from eggs began to decline when fine sediment levels in the substrate were in excess of 20 to 30 percent. Sediment deposition can result in surface water substrate armoring, trapping larvae as they attempt to emerge after hatching (Berry et al. 2003). Emergence success of cutthroat trout was reduced from 76 percent to 4 percent when fine sediment was added to spawning gravel redds (Weaver and Fraley, 1993).

Correlations have also been found between elevated suspended sediment and turbidity levels and reduced breeding and hatching success. Saunders and Smith (1965, as cited in Kerr 1995) found that increased suspended sediment levels reduced fish spawning activity. Other studies document lower survival rates for eggs spawned in surface waters with elevated suspended sediment and turbidity levels (Chapman 1988, as cited in Canadian DFO 2000). Reynolds (1989, as cited in Henley et al. 2000) found that increases in turbidity levels increased sac fry mortality in arctic grayling (*Thymallus arcticus*).

2.3.3.4 Habitat Loss

Like invertebrates, fish use crevices in waterbody beds for feeding, shelter from predators and high flow events, and reproduction. Loss of these interstitial spaces degrades fish habitat (USEPA 2006b). Sediment can also decrease the depth of or eliminate other important habitats such as stream pools and wetlands adjacent to surface waters. Riparian wetlands are particularly important for fish spawning. Fish are also affected by loss or decline in aquatic macrophytes, which are important sources of shelter and food for a number of fish species. Coral reef bleaching and decline severely impact fish habitat in coral reef ecosystems.

2.3.3.5 Juvenile Growth and Survival

High sediment concentrations can impact juvenile fish more severely than adult fish. Smallmouth bass reduced growth after a 24-hour exposure to suspended sediment levels as low as 11.4 mg/L (Berry et al. 2003). Reynolds et al. (1989, as cited in Bilotta and Brazier 2008) found 6 to 15 percent mortality of arctic grayling sac fry after 24 hours of exposure to suspended sediment levels of 25 to 65 mg/L. Mortality of 41 percent was documented after 72 hours of exposure to suspended sediment levels of 185 mg/L. Juvenile coho and Chinook salmon exhibited abnormal surfacing behavior under elevated suspended sediment concentrations, rendering them more vulnerable to avian predators (Canadian DFO 2000). This study also notes that excessive sediment levels can stress juvenile fish and consequently increase their predation risk. These types of effects can reduce the strength of a year class (Berry et al. 2003).

2.3.3.6 Physical Damage

At high levels, suspended sediment, particularly more angular particles, are capable of causing physical damage to fish by clogging gills (which impairs breathing) and by skin abrasion (Bell 1973, as cited in Waters 1995). Kerr (1995) notes that fish can withstand short episodes of high sediment concentrations by producing mucus to protect skin and gills, but this reaction stresses fish. Excessive gill mucus can cause gill tissue traumatization and asphyxiation. The severity of damage appears to be related to sediment dose, exposure duration, and particle size and angularity. Fish can release stress hormones (i.e., cortisol and epinephrine) in response to decreased gill function. Reid et al. (2003) finds that elevated sediment levels reduces the ability of rainbow trout to withstand hypoxic conditions and thereby reduces their average life span.

2.3.4 Other Wildlife Dependent on Aquatic Ecosystems

Birds, mammals, reptiles, and amphibians that consume aquatic plants, invertebrates, fish, and other aquatic organisms or otherwise utilize aquatic habitats for shelter and reproduction can also be affected by elevated sediment and turbidity levels in surface waters. Some species are sufficiently mobile that they can avoid impacted aquatic communities and seek substitutes, if available and accessible (Berry et al. 2003).

Welsh and Ollivier (1998) documented lower densities of two salamander species and one frog species in streams impacted by sedimentation from a construction site. The authors postulated that sedimentation of interstitial spaces in the stream’s substrate was the cause because of the organisms’ dependence on these spaces. Interstitial spaces provide amphibians with shelter from predators and high flows and habitat for food (e.g., diatoms and periphyton) production. Concerns have been raised about the potential impact of sedimentation on the endangered Barton Springs salamander (*Eurycea sosorum*), which depends on a coarse substrate and healthy aquatic macrophyte population (USFWS 2005).

Sedimentation can also affect or eliminate riparian wetlands, important habitats for the laying of amphibian egg masses (USEPA 2006b).

There are few studies available on the effects of elevated sediment and turbidity levels in aquatic ecosystems on fundamentally terrestrial wildlife that utilize aquatic ecosystems. Most available studies examine effects on birds. Loons appear to need clear water for hunting fish and may avoid turbid waters for nesting. Other studies documented birds modifying their fish hunting behaviors and distribution on a surface water in order to avoid more turbid areas, probably because of increased foraging difficulty (Berry et al. 2003).

2.3.5 Threatened and Endangered Species

Threatened and endangered (T&E) and other special status species can be adversely affected by elevated turbidity and sediment levels. The multiple ways in which elevated sediment and turbidity levels impact a variety of organisms are discussed above. These impacts can reduce organism growth, health, survival, and reproduction, thereby leading to further decline in an impacted T&E species population. The potential further decline of listed species or critical habitat for these species from construction activity is particularly important because these species are already rare and at risk of irreversible decline.

The federal trustees for T&E species are the Department of the Interior's U.S. Fish and Wildlife Service (USFWS) and the Department of Commerce's National Marine Fisheries Service (NMFS). USFWS is responsible for terrestrial and freshwater species (including plants) and migratory birds, and NMFS deals with marine species and anadromous fish (USFWS 2008). Various state agencies and departments have state-level jurisdiction over T&E species and habitats of concern.

A species is federally listed as "endangered" when it is likely to become extinct within the foreseeable future throughout all or part of its range if no immediate action is taken to protect it. A species is listed as "threatened" if it is likely to become endangered within the foreseeable future throughout all or most of its range if no action is taken to protect it. The 1973 Endangered Species Act outlines detailed procedures used by the Services to list a species, including listing criteria, public comment periods, hearings, notifications, time limits for final action, and other related issues (USFWS 1996).

A species is considered to be federally threatened or endangered if one or more of the following listing criteria apply (USFWS 2007):

- The species' habitat or range is currently undergoing or is jeopardized by destruction, modification, or curtailment.
- The species is overused for commercial, recreational, scientific, or educational purposes.
- The species' existence is vulnerable because of predation or disease.
- Current regulatory mechanisms do not provide adequate protection.
- The continued existence of a species is affected by other natural or manmade factors.

States and the federal government have also included species of "special concern" on their lists. These species have been selected because they are (1) rare or endemic, (2) in the process of being listed, (3) being considered for listing in the future, (4) found in isolated and fragmented habitats, or (5) considered a unique or irreplaceable state resource.

The federal government and individual states develop and maintain lists of species that are considered endangered, threatened, or of special concern. The federal and state lists are not identical: a state does not list a species that is on the federal list if it is extirpated in the state. States may also list a species that is not on the federal list if the species is considered threatened or endangered at the state, but not federal, level.

Information on federally listed T&E species is available from the USFWS Threatened and Endangered Species System (TESS) database (USFWS 2009), available at http://www.fws.gov/endangered/wildlife.html. Information on both federal and state listed species is available online in the NatureServe database (NatureServe 2009) at http://www.natureserve.org/explorer/. Additional information on state-listed species is available from state T&E species coordinators.

Numerous physical and biological stressors have resulted in the listing of multiple aquatic species as threatened or endangered. Major factors include habitat destruction or modification, displacement of populations by exotic species, dam building and impoundments, various point and nonpoint sources of pollution, poaching, and accidental deaths due to human activity.

Because construction activity occurs in every state and in or near a variety of waterbody types, there are many potentially affected T&E aquatic species. The current USFWS list of T&E species includes 255 aquatic species in five species groups: fish, amphibians, mollusks, crustaceans, and corals (USFWS 2009). For a complete list of these species, see *Appendix B.* This list is only a partial representation of potentially affected species because it does not include aquatic or aquatic ecosystem-dependent plants, insects, arachnids, reptiles, birds, and mammals. Although the USFWS list of T&E species does not provide information on causes of endangerment for each species, a review of the ecological literature suggests that elevated sedimentation, turbidity, and suspended sediment levels may have adverse impacts on species that are already in peril either directly or indirectly through impacts on supporting food chains (Cloud 2004; NatureServe 2009; USFWS 2009).

Several T&E species are thought to be particularly susceptible to excessive sedimentation, turbidity, and suspended sediment levels. One such example is *Amblema neislerii*, an endangered freshwater mussel commonly known as the fat threeridge. As the NatureServe Explorer database notes, "…the species and its habitats continue to be impacted by excessive sediment bed loads of smaller sediment particles, changes in turbidity, [and] increased suspended solids…" (NatureServe 2009). Other similarly sediment-susceptible species of mollusks include the oyster mussel (*Epioblasma capsaeformis*), purple bankclimber (*Elliptoideus sloatianus*), Louisiana Pearlshell (*Margaritifera hembeli*), Alabama Heelsplitter (*Potamilus inflatus*), and Ouachita rock pocketbook (*Arkansia wheeleri*) (USFWS 2009).

Many fish species are also vulnerable to sedimentation. Sedimentation is a risk factor for a number of shiner species, including the Arkansas river shiner (*Notropis girardi*), Pecos bluntnose shiner (*Notropis simus pecosensis*), and blue shiner (*Cyprinella caerulea*) (NatureServe 2009). The NatureServe Explorer notes that the biggest protection need for the threatened blue shiner (*Cyprinella caerulea*), native to the southeastern United States, is "…prevent[ing] siltation of habitat, especially during the spawning period…" (NatureServe 2009). Sedimentation is also listed as a threat factor to a number of minnow and darter species, such as the loach minnow (*Tiaroga cobitis*), spikedace (*Meda fulgida*), fountain darter (*Etheostoma fonticola*), leopard darter (*Percina pantherina*), watercress darter (*Etheostoma nuchale*), and vermilion darter (*Etheostoma chermocki*) (USFWS 1998, 2009; Drennen 2003; Cloud 2004; NatureServe 2009).

Sedimentation has also contributed to the threatened status of some populations of rainbow trout and several salmon species in the Pacific Northwest (NatureServe 2009). For example, California coho salmon and steelhead trout are listed as threatened and endangered by the state of California. The state has expressed concern about the impact of sediment discharges from construction sites on habitat throughout these species' range (McEwan and Jackson 1996; California Department of Fish and Game 2004). The state noted, for example, that discharges from the construction of Interstate 5 in California depleted spawning gravels for coho salmon (California Department of Fish and Game 2004).

Construction activities have been noted as a source of stress for the endangered Barton Springs salamander (*Eurycea sosorum*) (USFWS 2005). The U.S. Fish and Wildlife Service (2004) also issued a biological opinion addressing potential harm to the Arkansas river shiner (*Notropis girardi*) from a bridge construction project in Texas. While USFWS concluded that the project would not jeopardize the continued existence of the species, it did note that the river's fish community would be adversely affected by temporary loss of habitat, seining and handling of individual fish necessary to remove them from the immediate vicinity of the construction site, increased turbidity in the river, and harassment due to construction activity (e.g., equipment usage, materials storage, foot and vehicle traffic, installation of sediment and erosion controls, incidental deposition of debris in river). USFWS stated that these impacts might affect habitat access and seasonal movement of the fish.

Texas has identified the threatened and endangered aquatic plants Texas wild-rice (*Zizania texana*) and puzzle sunflower (*Helianthus paradoxus*) as vulnerable to siltation. Texas wild-rice also grows best in water with little or no turbidity. Little Aguja pondweed (*Potamogeton clystocarpus*) was identified as vulnerable to substrate texture modification (USFWS 1998).

2.4 Sediment and Turbidity Impacts on Human Use of Aquatic Resources

As sediment eroded from construction sites settles in surface waters or elevates sediment concentrations and turbidity in the water column, human uses of surface waters and human-built elements of the environment can be damaged. Damage from discharge of sediment and turbidity to surface waters has been recognized in the literature for several decades (Wolman and Schick 1967). Impacts to several types of human use of aquatic resources are described below.

2.4.1 Navigation on Surface Waters

Navigable waterways, including rivers, lakes, bays, shipping channels, and harbors, are an integral part of the United States' industrial transportation network. Navigable channels are prone to reduced functionality due to sediment build-up, which can reduce the navigable depth and width of the waterway (Clark et al. 1985). Increased sedimentation can lead to increased navigational difficulties and inefficiencies such as groundings and delays (Osterkamp et al. 1998). Shipping companies may switch to lighter loads or smaller vessels, which are generally less efficient. Removing sediment to keep navigable waterways passable requires dredging, which can be costly. The U.S. Army Corps of Engineers (USACE) spends an average of more than $572 million (2008$) every year to dredge waterways and keep them passable (USACE 2009).

Dredging is itself an environmentally disruptive activity because it significantly disturbs the physical structure of a surface water's bed and because dredged material may contain significant quantities of toxic substances and heavy metals. Dredging can disturb contaminants that have settled to the bottom of the waterway, increasing the potential for their uptake by fish and other aquatic organisms. Dredged sediment may be disposed of in another section of the waterbody or watershed, relocating the problem rather than removing it. Additionally, if the sediment removed from a site is contaminated, it can add to pollution at the disposal site (Clark et al. 1985).

In addition, unless there is overdredging to compensate or sedimentation is monitored so that dredging activity may be timed optimally, waterbodies will be on average be less navigable than if sedimentation rates were reduced.

In areas with significant sediment contamination, dredging may not be a feasible option because of high disposal costs for the dredged sediment. This can lead to damage of shipping vessels or shipping inefficiencies. An example is the Great Lakes Harbor and Indiana Harbor Ship Canal, where navigational dredging has not been conducted since 1972 because of the difficulty of contaminated dredge spoil disposal. Ships have trouble navigating the harbor and canal and must enter the harbor while loaded at less than optimum vessel drafts. There is also restricted use of some docks, requiring unloading at alternative docks and double handling of bulk commodities to the preferred dock. In 1995, the increased transportation costs associated with the lack of dredging were estimated to be $12.4 million annually (USEPA 2004c).

Chapter 7 of this report describes the methodology for and presents the results of EPA's analysis of benefits to navigation from several alternative regulatory options.

2.4.2 Reservoir Water Storage Capacity

Reservoirs are water impoundments, often manmade, that serve many functions, including providing drinking water, flood control, hydropower supply, and recreational opportunities. The National Inventory of Dams includes more than 75,000 dams (Renwick et al. 2005). Renwick et al. (2005) estimated the presence of 375,000–750,000 additional manmade impoundments in the United States, of which the majority are smaller than 1 acre in area. These smaller impoundments are used for recreation, livestock water supply, sediment trapping and erosion control, and other purposes. Sediment in streams can be carried into reservoirs and smaller impoundments, where it can settle and build up layers of silt over time.

Historically, the United States Geological Survey (USGS) has recorded an average of 2.6 billion pounds of sediment deposition settles in reservoirs each year (USGS 2007c). An increase in sedimentation rates will reduce reservoir capacity and utility. To replace this capacity, sediment must be dredged from reservoirs, or new reservoirs must be constructed (Clark et al. 1985). Both dredging and new reservoir construction can have a variety of environmental impacts. Crowder (1987) estimated that the United States was losing about 0.22 percent of its reservoir capacity each year due to sedimentation. Clark et al. (1985) noted that total U.S. reservoir capacity is filling up slowly and has enough excess capacity dedicated to hold sediment build-up over hundreds of years. However, the study went on to conclude that while total reservoir sedimentation is manageable, sedimentation is far from uniform and that, in approximately 15 percent of U.S. reservoirs, sedimentation rates exceeded 3 percent of capacity annually and, in the more extreme cases, 10 percent per year (Clark et al. 1985).

Sediment can also affect reservoir evaporation rates. Turbid waters tend to be sharply stratified, with a warm layer at the surface and a cooler layer below. Because warm water evaporates faster, turbidity can cause higher rates of water loss from reservoirs. However, turbidity may also increase the reflectivity of water, which makes water absorb solar heat more slowly than it otherwise would and reduces evaporation rates (Clark et al. 1985). The overall effect from temperature stratification in an individual reservoir could be positive or negative.

Wolman and Schick (1967) describe the loss of the use of a Massachusetts reservoir because of sediment discharged from airport construction. Significant resources were expended over two successive years for supply and pumping of water from alternative sources.

Chapter 8 of this report describes the methodology for and presents the results of EPA’s analysis of benefits from reduced sedimentation of larger reservoirs under several alternative regulatory options.

2.4.3 Municipal Water Use

Discharges from construction sites can affect the quality and cost of providing drinking water. Sediment, turbidity, and other discharged pollutants negatively affect water quality and require increased spending on treatment measures such as settlement ponds, filtration, and chemical treatment (Osterkamp et al. 1998). There is an additional cost associated with removing sludge that is created during the treatment process (USEPA 2007b). The presence of pollutants or dissolved minerals in drinking water may also affect the flavor and odor of water.

Surface water sedimentation can impede water flow into drinking water treatment facility intakes. Facilities may need to expend resources to unblock intakes or rebuild them in a different part of the drinking water supply surface water.

Construction site discharge impacts on municipal water supplies have been documented in the literature. Wolman and Schick (1967) noted that erosion and deposition of sediment had compromised use of surface waters as municipal water supplies in Maryland. They stated that where storage facilities were lacking and water was pumped directly from a river, sediment concentration spikes following rain events could force facilities to temporarily cease water withdrawals. This practice was practicable for only a very short period of time. Highway officials in the state had compensated municipalities for the construction of additional water storage facilities and for changing water intake locations to cope with sediment discharges from highway construction sites. Tan and Thirumurthi (1978) documented increases in turbidity, suspended solids, nitrogen, and conductivity levels in water supply lakes due to highway construction in Nova Scotia, Canada. Buckner (2002) described contamination of a municipal water supply due to runoff from an upstream highway construction site.

Chapter 9 of this report describes the methodology for and presents the results of EPA's analysis of benefits from reduced impacts to municipal water supplies under several alternative regulatory options.

2.4.4 Industrial Water Use

Elevated sediment and turbidity levels may have negative effects on industrial water users. Suspended sediment increases the rate at which hydraulic equipment, pumps, and other equipment wear out, causing accelerated depreciation of capital equipment. Sediment can also clog water intake systems at power plants and other industrial facilities and possibly require their relocation to another part and/or depth level of the surface water. Some industrial facilities treat water before use, and elevated sediment and turbidity levels may require additional treatment (Osterkamp et al. 1998) or make a surface water source unusable.

Wolman and Schick (1967) noted that erosion and deposition of sediment had led to turbid waters unsuitable for industrial uses in the state of Maryland. They stated that even small amounts of sediment can cause problems for industrial operations such as vegetable processing or cloth manufacture. They also noted that sediment had led to failure of pumping equipment. Excess sediment levels may require the use of filters to improve water quality.

At least one positive effect from elevated turbidity levels is also possible, however. Since turbidity may reduce the rate at which waterbodies absorb solar heat, more turbid waterbodies may supply cooler water, which in turn could improve the efficiency of cooling water systems at power plants and other industrial facilities (Clark et al. 1985).

2.4.5 Agricultural Water Use

Irrigation water that contains sediment or other pollutants from construction sites can harm crops and reduce agricultural productivity. Suspended sediment can form a crust over a field, reducing water absorption, inhibiting soil aeration, and preventing emergence of seedlings. Sediment can also coat the leaves of plants, inhibiting plant growth and reducing crop value and marketability. Furthermore, irrigation water that contains dissolved salts or pollutants can harm crops and damage soil quality (Clark et al. 1985).

Wolman and Schick (1967) noted that erosion and deposition of sediment in surface waters in the state of Maryland had led to failure of pumping equipment. Pumps are often used for the movement of irrigation water. Excess sediment levels may require the use of filters to improve water quality.

Surface water sedimentation can block irrigation water intake structures or require their relocation to another part of the surface water. Sedimentation can also cause sediment build-up in irrigation water canals and other transport systems (Osterkamp et al. 1998), reducing their efficiency. Sediment can also accumulate in livestock water supply impoundments and reduce their capacity (see *Section 2.4.2*).

2.4.6 Stormwater Management and Flood Control

Sediment in discharges from construction sites can clog or fill ditches, culverts, storm sewer pipes and basins, stormwater detention ponds, infiltration basins, and other stormwater management structures (Osterkamp et al. 1998). Sedimentation also decreases the capacity of natural stream and river channels, ponds, lakes, and reservoirs, increasing the likelihood of overbank flow events and their magnitude (Paul and Meyer 2001). Wolman and Schick (1967) noted that erosion and deposition of sediment in the state of Maryland had led to sediment deposition in streams and overflow and clogging of storm drains.

If left in place, sedimentation can increase flooding. The U.S. Army Corps of Engineers (2007) stated that channel sedimentation due to construction discharges contributed to flooding of a Virginia residential neighborhood. Preventing flood damages from excessive sedimentation may require increased maintenance efforts such as dredging (Clark et al. 1985), vacuuming, and other types of sediment removal from stormwater management structures and surface waters. If sedimentation removal is performed in surface waters or in structures directly adjacent to surface waters, it can create additional environmental impacts such as resuspension of sediments and associated turbidity and contaminants in the water column and disturbance of the physical structure of the surface water bed and the associated benthic aquatic community.

Surface water and stormwater management system sedimentation can increase the severity of property damages to bridges, roads, farmland, and other private and public property from flooding. Additionally, sediments carried by flood waters can damage property and can be expensive to remove once deposited, particularly in developed areas (Clark et al. 1985; Osterkamp et al. 1998). Clark et al. (1985) estimated flooding damages attributable to *all* sediment discharges to be $1.5 billion (2008$), annually.

The quantity of sediment captured in stormwater control structures is unknown or could be very variable. Nelson and Booth (2002) did not observe significant long-term sediment accumulation in the stormwater retention/detention facilities (including the ditches and pipes connecting them to surface waters) of newer residential developments in Issaquah, Washington. They did note, however, that a significant amount of sediment was removed every year from catch-basins in the city of Issaquah, though a large fraction of this sand may have derived from winter road sanding.

2.4.7 Recreational Uses and Aesthetic Value

Polluted water greatly reduces the aesthetic appeal of a variety of recreational activities that take place in or near surface waters, including boating; swimming; hunting; and outings to hike, jog, picnic, camp, and view wildlife. Turbidity and suspended sediments are highly visible pollutants. Murky and visually unpleasant water and odors can detract from the enjoyment gained through outdoor activities. Sedimentation of streams, reservoirs, lakes, and bottomlands can reduce their depth and thus their capacity for boating and swimming (Osterkamp et al. 1998).

Turbidity caused by sediment discharges may affect the safety of boating. Turbidity may obscure underwater obstacles, making collisions more likely. Increased sedimentation levels may create sandbars, increasing the chances of running aground. Clark et al. (1985) estimated that turbidity (from all sources) may be responsible for as many as 200 boating fatalities and many more injuries each year. Turbidity may also create safety hazards for swimmers by reducing the ability to see underwater hazards, increasing the frequency of diving accidents by impairing the ability to gauge water depth, and making location of swimmers in danger of drowning more difficult.

Wolman and Schick (1967) noted that erosion and deposition of sediment in the state of Maryland had damaged recreation areas. Concern about potential highway construction impacts to aesthetic and recreational values of a small lake was sufficient to motivate residents of an adjoining neighborhood to initiate monitoring of lake water quality (Line 2009).

Aesthetic degradation of land and water resources resulting from sediment and turbidity discharges can also reduce the market value of property near surface waters and thus affect the financial status of property owners. For example, a hedonic price study by Bejranonda et al. (1999) found that "the rate of sediment inflow entering the lakes has a negative influence on lakeside property rent." Sediment discharges also have a significant impact on stream morphology. For example, higher coarse-sediment load leads to an increase in width of the river bed and, as a result, bank erosion (Wheeler et al. 2003). A 1993 study of Lake Erie's housing market found that "erosion-prone lakeshore property will be discounted" (Kriesel et al. 1993). Stabilization of stream banks leads to an increase in the value of surrounding property (Streiner and Loomis 1996).

2.4.8 Recreational and Commercial Fishing

Pollutants can negatively affect local flora and fauna, negatively impacting aesthetic appeal, wildlife viewing, and hunting for game (Osterkamp et al. 1998). By harming fish and shellfish communities, sediment can reduce fish and shellfish numbers or cause more desirable fish and shellfish to be replaced by less desirable fish and shellfish in a given location.

As discussed in *Section 2.3*, sediment, turbidity, and other discharges from construction sites can reduce fish and shellfish populations by inhibiting their reproduction, growth, and survival. In some areas, desirable sediment-sensitive fish may be replaced by less-desirable, sediment-tolerant fish and shellfish. These population changes and reductions would reduce the size of commercial and recreational harvest by lowering both the total abundance of organisms and their individual size. These changes negatively affect recreational anglers, subsistence anglers, commercial anglers, fish and shellfish sellers, and consumers of fish and shellfish products.

In addition, sediments, turbidity, and other pollutants reduce the aesthetics of a waterbody, which can reduce anglers' enjoyment of their fishing experience and their choices of how often and where to fish. Sediment and turbidity may also affect recreational anglers by reducing the distance over which fish can see lures, resulting in lower catch rates (Clark et al. 1985).

2.5 Sediment and Turbidity Criteria

Natural levels of sediment and turbidity play important roles in aquatic ecosystems, providing habitat for benthic species, protection from predators, nutrient transport, and other functions. Appropriate levels of sediment are waterbody specific, as sediment levels vary naturally among different types of waterbodies according to geology, topography, stream gradient, waterbody morphology, vegetative land cover, climate, soil erodibility, and other landscape characteristics of the contributing watershed (USEPA 2006b). When sediment and turbidity enter surface water at elevated levels, they can raise TSS and turbidity levels in the water column, increase deposition of sediment on the waterbody's bed, and degrade overall water quality (USEPA 2006b). Elevated sediment and turbidity levels also negatively impact human use of aquatic resources for recreation, drinking water supply, agricultural use, industrial use, and navigation and can impair stormwater management system function and surface water aesthetics.

EPA and many states have issued criteria that describe appropriate sediment and turbidity levels for surface waters. Many of these criteria are narrative and provide a qualitative description of appropriate levels. Other criteria are numeric.

2.5.1 Federal Sediment Criteria

In 1987, EPA published the following guidance for developing numeric water quality criteria for sediment and turbidity:

> Solids (Suspended, Settleable) and Turbidity – Freshwater fish and other aquatic life: Settleable and suspended solids should not reduce the depth of the compensation point for photosynthetic activity by more than 10 percent from the seasonally established norm for aquatic life (USEPA 1987).

This criterion has not been widely adopted by states. Guidance addressing primarily aesthetic properties, however, has been adopted by several states.

> Aesthetic Qualities – All waters shall be free from substances attributable to wastewater or other discharges that: settle to form objectionable deposits; float as debris, scum, oil, or other matter to form nuisances; produce objectionable color, odor, taste, or turbidity; injure or are toxic or produce adverse physiological response in humans, animals, or plants; [or] produce undesirable or nuisance aquatic life (USEPA 1987).

EPA has not issued new sediment or turbidity criteria since 1987. However, EPA has published a document providing guidance on setting sediment criteria: *Framework for Developing Suspended and Bedded Sediments (SABS) Water Quality Criteria* (USEPA 2006b).

2.5.2 State Sediment Criteria

Many states have developed their own criteria to designate appropriate sediment levels for waters within the state. Some criteria vary by waterbody type. Streams with hard substrates (e.g., gravel, cobble, bedrock) or cold water fisheries typically have more stringent criteria than streams with soft substrates (e.g., sand, silt, clay) or warm water fisheries. Hawaii has separate criteria for reefs (Berry et al. 2003). In addition, Total Maximum Daily Loads (TMDLs) addressing sediment and turbidity have been developed for some surface waters and contain information on acceptable levels specific to those waterbodies.

Many criteria are narrative statements describing the general nature of healthy sediment or turbidity levels without attempting to provide numeric guidelines. Narrative criteria most frequently address turbidity or surface water appearance (e.g., “free of substances that change color or turbidity”). Other criteria refer to undesirable biological effects (e.g., “no adverse effects” or “no actions which will impair or alter the communities”) (Berry et al. 2003). EPA (USEPA 2006b) provides a summary of state narrative and numeric criteria (though some criteria may have been modified since this information was compiled).

A number of states have also set numeric criteria for TSS or turbidity levels. Most of these numeric criteria are for turbidity. Turbidity criteria exist as absolute values or ranges of values or as a permitted exceedance beyond background turbidity levels. Some states have issued numeric criteria for suspended sediments.

A number of states have criteria based on sedimentation levels over a time period or during a storm event. Values are typically 5 mm during an individual event (e.g., during the 24 hours following a heavy rainstorm) for streams with hard substrate bottoms and 10 mm for streams with soft bottoms. Hawaii’s reef criterion is 2 mm of deposited sediment after an event (Berry et al. 2003).

Table 2-3 summarizes state suspended sediment and turbidity criteria for surface water quality. States using narrative criteria are noted with a “Yes” in the “Narrative TSS” or the “Narrative Turbidity” column. Values for numeric criteria and some details on their implementation are provided in the “Numeric TSS” and “Numeric Turbidity” columns. This table is intended to provide an overview of state numeric suspended sediment and turbidity water quality requirements and does not summarize all details relevant to their applicability. In addition, states periodically update their water quality criteria. Readers should consult individual state water quality criteria publications in order to obtain the most complete and current requirements.

Table 2-3: Suspended Sediment and Turbidity Criteria for Surface Water Quality by State

<table>
<tr><th>State</th><th>Numeric TSS (mg/L)</th><th>Narrative TSS</th><th>Numeric Turbidity Criteria</th><th>Notes on Numeric Turbidity Criteria</th><th>Narrative Turbidity</th></tr>
<tr><td>Alabama</td><td>-</td><td>-</td><td><50 NTU increase</td><td>Source: Alabama DEM (2009).</td><td>Yes[1]</td></tr>
<tr><td>Alaska</td><td>-</td><td>-</td><td><5 NTU increase[2]
<10% increase (max. 25 NTU)[3]</td><td>Source: Alaska DEC (2003)</td><td>Yes</td></tr>
<tr><td>Arizona</td><td>-</td><td>-</td><td>10-50 NTU[4]</td><td>Human Contact: 50 NTU in rivers, 25 NTU in lakes
Cold Water Fishery: 10 NTU
Warm Water Fishery: 50 NTU in rivers, 25 NTU in lakes</td><td>Yes</td></tr>
<tr><td>Arkansas</td><td>-</td><td>-</td><td>10-75 NTU[4] for baseflow values</td><td>“Non-point source runoff shall not result in the exceedance of the in stream all flows values in more than 20% of the ADEQ ambient monitoring network samples taken in not less than 24 monthly samples."
Source: Arkansas Pollution Control and Ecology Commission (2007).</td><td>Yes</td></tr>
<tr><td>California</td><td>-</td><td>-</td><td><20% increase[2]
<10% increase[3]</td><td>20% increase applies to Central Coast Region and is measured in JTU
10% increase applies to San Francisco Region
Source: California EPA 2009a, b</td><td>Yes</td></tr>
<tr><td>Colorado</td><td>-</td><td>Yes</td><td>-</td><td></td><td>-</td></tr>
<tr><td>Connecticut</td><td>-</td><td>-</td><td><5 NTU increase</td><td>Under ambient conditions. Class AA criteria
Source: Connecticut DEP (2002).</td><td>-</td></tr>
<tr><td>Delaware</td><td>-</td><td>-</td><td><10 NTU increase</td><td>For all fresh waters and mixing zones</td><td>-</td></tr>
<tr><td>District of Columbia</td><td>-</td><td>-</td><td>-</td><td></td><td>Yes[5]</td></tr>
<tr><td>Florida</td><td>-</td><td>-</td><td><29 NTU increase</td><td></td><td>Yes</td></tr>
<tr><td>Georgia</td><td>-</td><td>Yes</td><td>-</td><td></td><td>Yes</td></tr>
<tr><td>Hawaii</td><td>10-55[7]</td><td>Yes</td><td>2-25 NTU for streams
0.1-15 NTU for coastal/marine waters</td><td>Turbidity levels not to exceed NTU as specified below:
<table>
<tr><th rowspan="2">Water type</th><th rowspan="2">Geometric mean</th><th colspan="2">More than</th></tr>
<tr><th>10% of the time</th><th>2% of the time</th></tr>
<tr><td>Stream (wet season)</td><td>5</td><td>15</td><td>25</td></tr>
<tr><td>Stream (dry season)</td><td>2</td><td>5.5</td><td>10</td></tr>
<tr><td>All Estuaries</td><td>1.5</td><td>3.0</td><td>5.0</td></tr>
<tr><td>Pearl Harbor</td><td>4.0</td><td>8.0</td><td>15.0</td></tr>
<tr><td>Embayment</td><td>0.4</td><td>1.0</td><td>1.5</td></tr>
<tr><td>Open Coastal (wet season)</td><td>0.5</td><td>1.25</td><td>2.0</td></tr>
<tr><td>Open Coastal (dry season)</td><td>0.02</td><td>0.05</td><td>1.0</td></tr>
<tr><td>Oceanic</td><td>0.03</td><td>0.1</td><td>0.2</td></tr>
<tr><td>Marine</td><td>0.1</td><td></td><td></td></tr>
</table></td><td>Yes</td></tr>
</table>

Table 2-3: Suspended Sediment and Turbidity Criteria for Surface Water Quality by State

State	Numeric TSS (mg/L)	Narrative TSS	Numeric Turbidity Criteria	Notes on Numeric Turbidity Criteria	Narrative Turbidity
Idaho	-	Yes	<50 NTU increase at any time, <25 NTU increase over 10 consecutive days	For applicable mixing zones set by the department only *Source: Idaho DEQ (2008).*	Yes
Illinois		-	-		Yes
Indiana	-	Yes	-		Yes
Iowa	-	Yes	25 NTU		Yes
Kansas	-	Yes	-		Yes
Kentucky	-	Yes	-		Yes
Louisiana	-	Yes	25, 50, 150 NTU, or <10% increase[4]	25 NTU: freshwater lakes, reservoirs, and oxbows; designated scenic streams and outstanding natural resource waters 50 NTU: Amite, Pearl, Ouachita, Sabine, Calcasieu, Tangipahoa, Tickfaw, and Tchefuncte Rivers; estuarine lakes, bays, bayous, and canals 150 NTU: Atchafalaya, Mississippi, and Vermilion Rivers and Bayou Teche 10% Increase: All other waters	Yes
Maine	-	-	-		Yes
Maryland	-	-	150 NTU at any time 50 NTU monthly average		Yes
Massachusetts	-	Yes	-		Yes
Michigan	-	-	-		Yes
Minnesota	-	-	5, 10, or 25 NTU[4]	Domestic Consumption: Class A & B 5 NTU, Class C 24 NTU Fisheries & Recreation: Class A 10 NTU, Class B & C 25 NTU Industrial Consumption: Class A 5 NTU	-
Mississippi	-	Yes	50 NTU	Applicable to waters outside the limits of a 750-foot mixing zone	Yes
Missouri	-	Yes	-		Yes
Montana	-	Yes	-		Yes
Nebraska	-	-	-		Yes[7]
Nevada		Yes	Waterbody specific		Yes
New Hampshire	-	Yes	0-10 NTU increase[4]	Class A waters: No turbidity other than natural Class B & C: 10 NTU increase over natural	Yes
New Jersey	25	-	10, 30, or 50 NTU[4]	Freshwater: 50 NTU any time, 15 NTU 30-day average Saline water: 30 NTU any time, 10 NTU 30-day average Coastal saline: 10 NTU	-
New Mexico	-	Yes	<10 NTU increase[2] or <20% increase[3]	GA waterbody Turbidity shall not exceed 5 NTU *Source: New Mexico (2005)*	Yes

Table 2-3: Suspended Sediment and Turbidity Criteria for Surface Water Quality by State

State	Numeric TSS (mg/L)	Narrative TSS	Numeric Turbidity Criteria	Notes on Numeric Turbidity Criteria	Narrative Turbidity
New York	-	Yes	Waterbody specific		Yes[8]
North Carolina	500[9]	Yes	10, 25, or 50 NTU[4]	10 NTU in trout waters 25 NTU in other lakes and reservoirs 50 NTU in other streams If background exceeds these levels, no increase	-
North Dakota	30	-	-		-
Ohio		-	-		-
Oklahoma	-	-	10 or 50 NTU[4]	Cold water aquatic communities/trout fisheries: 10 NTU; Lakes: 25 NTU; Other Surface Waters: 50 NTU If background exceeds these levels, point sources may not cause increases above ambient levels	-
Oregon	-	Yes	<10% Increase	Exception for temporary increases to do "legitimate activities" including dredging and construction, providing all practicable turbidity control techniques are applied.	Yes
Pennsylvania	-	Yes	40-100 NTU	Specific limits only apply to Neshaminy Basin	Yes
Puerto Rico	-	-	Specific to waterbody class	Class SB shall not exceed 10 NTU, except by natural causes; Class SC shall not exceed 10 NTU; Class SD shall not exceed 50 NTU, except when due to natural phenomena *Source: Puerto Rico EQB (2003).*	Yes
Rhode Island	-	-	5 NTU or <10 NTU increase[4]	Class A: 5 NTU Class B & C: 10 NTU increase	-
South Carolina		Yes	10 NTU or <10% increase	Providing existing uses are maintained Specific to freshwaters suitable for supporting trout stocks	Yes
South Dakota	53-263 (at any time); 30-150 (monthly average)[4]	Yes	-		Yes
Tennessee	-	Yes	-		Yes
Texas	-	Yes	-		Yes
Utah	35-90[4]	Yes	10 or 15 NTU[4]	10 NTU for cold and warm water game fish 15 NTU for nongame fish, waterfowl, and other wildlife	Yes[10]
Vermont	-	-	10 or 25 NTU[4]	10 NTU for Class A(1) ecological waters and Class B cold water fish habitats 25 NTU for Class B warm water fish habitats	Yes
Virginia	-	Yes	-		Yes

Table 2-3: Suspended Sediment and Turbidity Criteria for Surface Water Quality by State

State	Numeric TSS (mg/L)	Narrative TSS	Numeric Turbidity Criteria	Notes on Numeric Turbidity Criteria	Narrative Turbidity
Washington	-	-	5 or 10 NTU[2] or <10% or <20% increase[3 4]	Class AA &A: 5 NTU[2] or 10% increase[3] Class B & C: 10 NTU[2] or 20% increase[3] Lakes: 5 NTU increase Also includes a flow-based criteria based on flow from construction site. Flows ranging from 10 to above 100 CFS have turbidity measured at 100 to 300 feet from downstream activity *Source: Washington (1997)*	-
West Virginia	-	-	10 NTU[2] or <10% increase[3]		-
Wisconsin	-	Yes	-		Yes
Wyoming	-	Yes	<10 or 15 NTU increase[4]	<10 NTU increase for Class 1 and 2 waters that are cold water fisheries <15 NTU increase for Class 1 and 2 waters that are warm water fisheries and all Class 3 waters	Yes

1 "There shall be no turbidity of other than natural origin that will cause substantial visible contrast with the natural appearance of waters or interfere with any beneficial uses which they serve."
2 If naturally less than 50 NTU.
3 If naturally greater than 50 NTU.
4 Varies based on waterbody classification.
5 May not "produce objectionable odor, color, taste, or turbidity."
6 Applies during dry season only. Geometric mean of 10 mg/L, not exceeding 30 mg/L 10% of the time and 55 mg/L 2% of the time.
7 "To be aesthetically acceptable, waters shall be free from human-induced pollution which causes: 1) noxious odors; 2) floating, suspended, colloidal, or settleable materials that produce objectionable films, colors, turbidity, or deposits."
8 Waterbody types AA, A, B, C, D, SA, SB, SC, SD, I: no increase may cause substantial visible contrast from natural conditions.
9 Standard is for total dissolved solids (TDS).
10 For mixing zones.

Source: USEPA (2006b), unless otherwise noted.

2.6 Surface Water Quality Impairment from Sediment and Turbidity

Section 305(b) of the Clean Water Act (CWA) requires states, territories, and other jurisdictions of the United States to submit reports to EPA on the quality of their surface waters every two years. These entities have determined the appropriate uses of each waterbody within their jurisdiction. Uses can include recreation, drinking water source, navigation, cold water fishery, and wildlife habitat, among others. States and other entities determine appropriate narrative and/or numeric water quality criteria for each of the designated uses. The criteria describe the physical, chemical, and biological characteristics of a surface water able to fulfill its designated uses. The sediment criteria described in the section above are examples of such criteria. Typically, a surface water has criteria for multiple water quality parameters. If a waterbody fails to meet any one of its designated uses, CWA Section 303(d) requires a state or other entity to list the waterbody as “impaired.” If a waterbody meets its designated uses but is in danger of failing to do so in the future, the state or other entity must list the waterbody as “threatened” (USEPA 2005a).

The Assessment TMDL Tracking and Implementation System (ATTAINS) provides information on water quality conditions reported by the states to EPA under Sections 305(b) and 303(d) of the Clean Water Act. The information available in ATTAINS is updated as data are processed and are used to generate the biennial National Water Quality Inventory Report to Congress. This information reflects only the status of those waters that have been assessed. *Appendix A* provides information on the state water report year for which data were available for populating the tables below as of September 17, 2009.

According to ATTAINS, 49 percent of assessed reach miles (or 458,209 miles) have been identified as impaired; sediment contributes to impairment in 107,231 miles, and turbidity contributes to impairment in 26,278 miles. For lakes and reservoirs, 66 percent of assessed lake acres (or 11,545,337 acres) have been identified as impaired; sediment contributes to impairment in 715,558 lake acres, and turbidity contributes to impairment in 1,008,276 acres. For bays and estuaries, 63 percent of assessed square miles (or 11,222 square miles) have been identified as impaired; sediment contributes to impairment in 209 square miles, and turbidity contributes to impairment in 240 square miles. It should be noted that individual waters may be impaired by more than one pollutant. Although states tend to target their monitoring efforts to those surface waters they believe to be impaired, the total area of impaired surface waters due to sediment and turbidity is probably underestimated due to the low percentage of surface waters that were assessed. As of September 17, 2009, states had assessed only 26 percent of the nation’s reach miles, 42 percent of its lake acres, and 20 percent of its bay and estuary square miles.

Table 2-4 and *Table 2-5* present information on “sediment” and “turbidity and suspended solids” impairment by EPA Region as reported by the states in ATTAINS. This information is presented by EPA Region. *Figure 2-1* provides a map of the EPA Regions.

Table 2-4: Surface Waters Impaired by "Sediment," by EPA Region

EPA Region	Reach (miles)	Lake/Pond/Reservoir (acres)	Bay/Estuary (sq. miles)
1	236	443	4
2	1,123	39,277	0
3	8,188	193	12
4	11,126	23,333	–
5	34,410	57,965	–
6	3,567	153,478	193
7	9,659	144,125	–
8	9,597	147,592	–
9	198	–	–
10	29,127	149,152	–
Nation	**107,231**	**715,558**	**209**

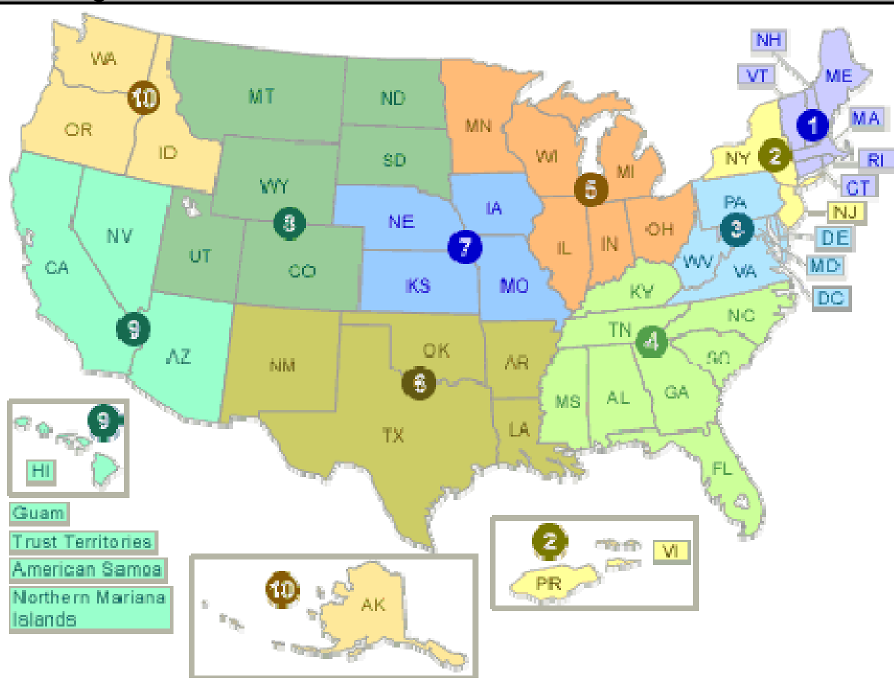
Source: EPA ATTAINS database (USEPA 2009a) as of 9/17/09.

Table 2-5: Surface Waters Impaired by "Turbidity" and "Suspended Solids," by EPA Region

EPA Region	Stream/ River (miles)	Lake/Pond/Reservoir (acres)	Bay/Estuary (sq. miles)
1	339	43,735	5
2	3,539	1,539	8
3	229	–	–
4	732	13,814	4
5	9,278	134,427	–
6	6,880	626,642	195
7	84	38,173	–
8	2,304	3,168	–
9	904	136,980	28
10	2,439	9,798	–
Nation	**26,728**	**1,008,276**	**240**

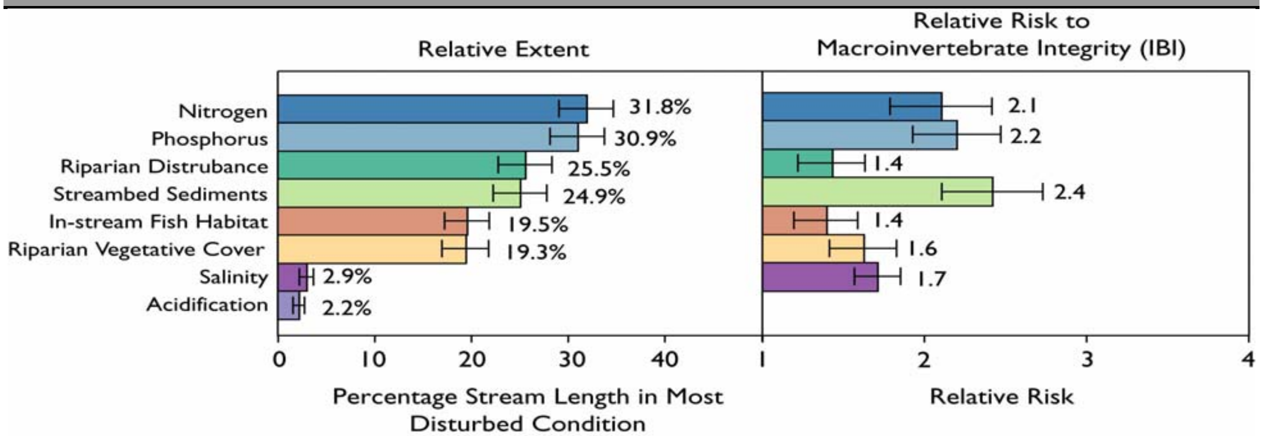
Source: EPA ATTAINS database (USEPA 2009a) as of 9/17/09.

Figure 2-1: EPA Regions



Water quality in the United States has also been assessed through a series of national, probability-based surveys known as the National Aquatic Resource Surveys. These surveys use randomized sampling designs, core indicators, and consistent monitoring methods and laboratory protocols to provide statistically defensible assessments of water quality at the national scale. The *Wadeable Streams Assessment* (USEPA 2006d) is a statistical survey of the smaller perennial streams and rivers that, according to the report, comprise 90 percent of all perennial stream miles in the United States. Excess streambed sedimentation is ranked as one of the most widespread stressors examined in the survey. (The survey did not analyze turbidity or suspended sediment levels.) According to the survey, 25 percent of streams have "poor" streambed sediment condition, and 20 percent are in "fair" condition relative to reference streams. The survey also examined the association between stressors and biological condition, and found that high levels of sediments more than double the risk for poor biological condition (see *Figure 2-2*).

Figure 2-2: Extent of Stressors and Their Relative Risk to the Biological Condition of the Nation's Streams



Source: USEPA (2006c).

Another National Aquatic Resource Survey, the *National Coastal Condition Report III* (USEPA 2008b), assesses coastal aquatic habitat and found water clarity (related to turbidity) to be poor in 17 percent of coastal waters. Data on excessive sedimentation was not collected for the report.

2.6.1 Current Total Suspended Solid Concentrations in U.S. Surface Waters

TSS concentrations vary from waterbody to waterbody due to differences in their contributing watersheds created by natural conditions and human activity. EPA used the SPARROW model to estimate current TSS concentrations in the United States' Reach File Version 1 (RF1) surface water network (see *Chapter 6* for more information). For this analysis, the RF1 network consists of approximately 650,000 miles of the largest rivers and streams in the coterminous United States and associated lakes, reservoirs, and estuarine waters. Due to data limitations, EPA did not model surface water turbidity levels. *Table 2-6* summarizes the distribution of TSS concentrations found in individual RF1 reaches as estimated by EPA's analysis. The median TSS concentration, weighted by reach length, is 302.9 mg/L. The range of TSS concentrations falls between 26.7 mg/L and 6,154.5 mg/L at the 5^{th} and 95^{th} percentiles, respectively. The average TSS concentration is 1,068.6 mg/L, indicating that concentrations in the 90^{th} to 95^{th}

percentile range are high enough to shift the average concentration significantly higher than the median concentration.

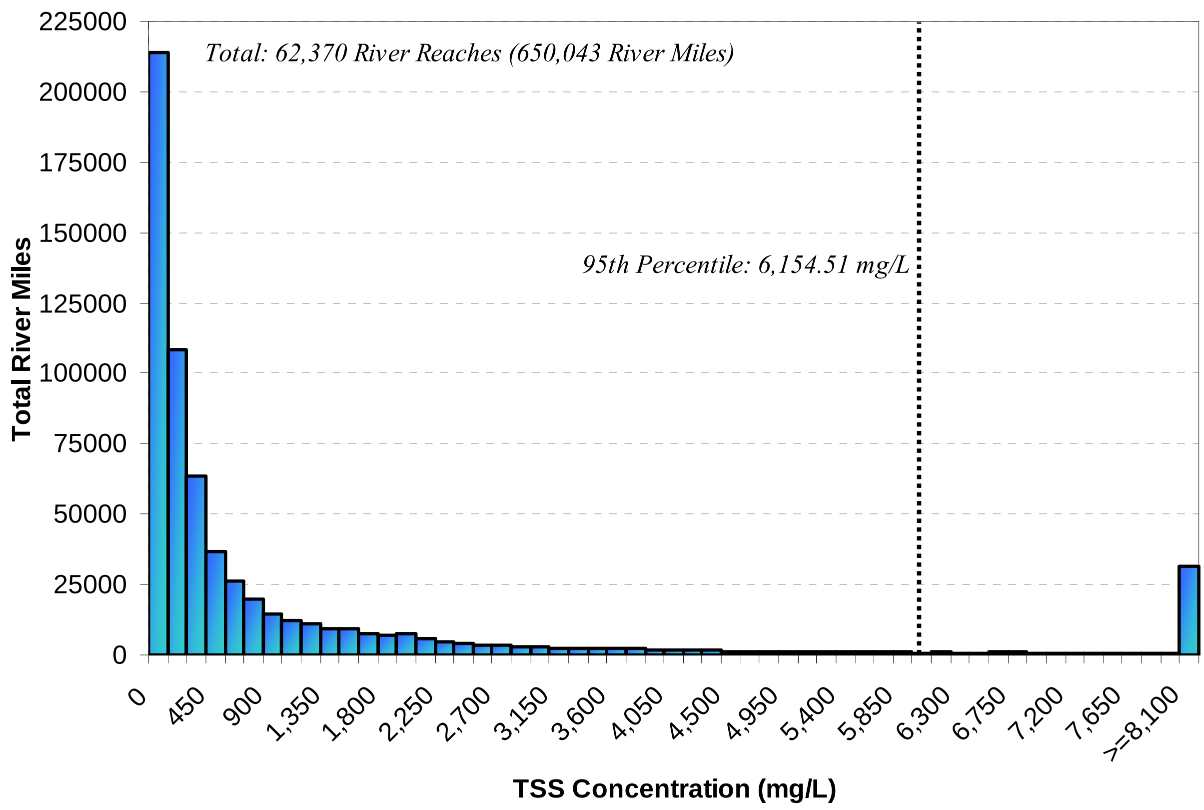
Table 2-6: SPARROW Distribution of TSS Concentrations in RF1 Reaches[1]

RF1 Reach Count	RF1 Reach Miles	Average TSS (mg/L)[1]	Distribution of TSS Concentrations in RF1 Reaches[1]				
			5th Percentile	25th Percentile	50th Percentile	75th Percentile	95th Percentile
62,370	650,043	1,068.6	26.7	102.9	302.9	1,079.6	6,154.5

[1] Concentrations weighted by reach length. Concentration estimates reflect replacement of potential outliers (defined as values above the 95th percentile) with the 95th percentile value.

Figure 2-3 shows total RF1 reach miles and their current TSS concentrations as predicted by SPARROW (not weighted by reach length). As shown below, the majority of waters have TSS concentrations below 1,000 mg/L, though substantially higher average concentrations up to and exceeding 6,000 mg/L can be found in some waters.

Figure 2-3: TSS Concentrations by Total RF1 Reach Miles as Predicted by SPARROW for Current Conditions



An important factor to consider when examining this information is that episodic precipitation events are the primary cause of construction site TSS discharges to surface waters. Most TSS discharge, therefore, takes place during or shortly after precipitation events. Once a precipitation event ceases, discharges from construction sites generally cease within a short time period. The estimates presented here, however, are not intended to reflect the higher surface water TSS concentrations associated with individual, episodic storm events. They instead represent average concentrations estimated to exist in RF1 reaches over multi-year time periods. EPA uses these long-term estimates because the data and modeling resources currently available to EPA do not permit the finer level of time resolution necessary to model individual storm events. A study by Wong (2005) documents that long-term averages of TSS concentration sampling data provide a different type of perspective on surface water conditions than data on observed 90^{th} and 98^{th} percentile concentrations, which better reflect the types of concentrations observed after precipitation or other erosion events.

EPA expects that, in general, surface water TSS concentrations during and immediately following storm events would be higher than the concentrations presented here. During periods when no precipitation event is available to cause a sediment discharge from a construction site, surface water sediment concentrations would generally be lower.

Current levels of sediment erosion and sediment and turbidity discharge to many surface waters in the United States are much higher than would be observed under natural, undisturbed conditions due to human activity. Sediment and turbidity levels in surface waters reflect these elevated levels of sediment input.

Human activities that have increased sediment erosion and transport in surface waters include agriculture, grazing, logging, construction, and mining. In addition, land use change has increased streambed erosion in many urbanized areas, mobilizing significant quantities of sediment (Nelson and Booth 2002; Renwick et al. 2005; Wilkinson 2005; Wilkinson and McElroy 2007). It is estimated that soil erosion due to human activity is approximately 10 times greater than erosion from all natural forces combined (Wilkinson 2005). Approximately one-third of all eroded sediment in the United States derives from agricultural activities (Osterkamp et al. 1998). Construction is considered to be another major source of erosion (Wilkinson 2005). Nelson and Booth (2002) documented a 50 percent increase in annual sediment yield due to human activity (primarily urbanization) in a western Washington watershed.

Only a portion of all eroded sediments are transported through the entire surface water system to the ocean in any given year (Renwick et al. 2005; Wilkinson 2005; Wilkinson and McElroy 2007). A large portion is stored on the land surface and in and adjacent to surface waters. Some surface waters are currently mobilizing historical sediments stored on floodplain terraces and in small impoundments (e.g., from past farming, mining, and clear-cutting practices). These stored sediments provide a large and continuing source of sediment input to the U.S. surface water network (Renwick et al. 2005).

3 Other Pollutants in Construction Site Discharges

In addition to mobilizing and discharging sediment and turbidity to surface waters, construction activity can discharge many other kinds of pollutants. Whereas sediment and turbidity are nearly ubiquitous in discharges from construction sites, the presence and quantity of other pollutants varies widely. These pollutants can derive from construction materials and equipment, historic site contamination, and natural soil and groundwater constituents. They may be carried in stormwater in solution or adsorbed to transported sediment particles. These "other pollutants," though not as well documented as sediment and turbidity in the construction context, can also impact water quality and therefore aquatic life and human use of water resources. This chapter discusses these pollutants and their potential sources, behavior in surface waters, and potential to impact aquatic organisms and human use of aquatic resources. *Section 3.1* provides an overview of the importance and concerns associated with these other pollutants. *Section 3.2* provides additional information on potential sources of pollutants on construction sites and changes in stormwater discharge. *Section 3.3* provides additional information on specific pollutants potentially found in construction site discharges. Pollutant-specific impacts from laboratory testing and field studies are cited as appropriate, but further information is available in *Chapter 4*, which provides a review and summary of studies from the literature that document specific cases of construction site discharges, the pollutants they contain, and their impacts on the aquatic environment.

3.1 Introduction

In addition to mobilizing and discharging sediment and turbidity to surface waters, construction activity can discharge many other kinds of pollutants. These pollutants can derive from construction materials and equipment, historic site contamination, and natural soil and groundwater constituents. Construction activity can also impact receiving waters by alteration of the hydrologic regime; for example, increasing the volume and intensity of stormwater runoff but decreasing a surface water's dry weather baseflow. These alterations may lead to erosion and sediment mobilization further down in the watershed of the construction activity. *Section 3.2* provides additional information on potential sources of pollutants on construction sites and changes in stormwater discharge.

The pollutants of interest include nutrients (nitrogen, phosphorus), organic compounds and materials (organic material, petroleum hydrocarbons, polycyclic aromatic hydrocarbons (PAHs), pesticides), metals, dissolved inorganic ions (major anions and cations), substances that can modify surface water pH, and pathogens (bacteria, fungi). Studies of construction sites have documented the presence of many of these pollutants in construction site stormwater, discharges, and receiving waters. *Section 3.3* describes their potential sources, their transport behavior, and their potential to impact the aquatic environment and resources.

3.1.1 Pollution Discharge and Transport Pathways

Most pollutants enter surface waters when precipitation erodes or dissolves materials and carries them to receiving waters. Many of the pollutants described in this section adsorb to or are a component of soil particles and travel with them as they erode (e.g., organic compounds, metals, nutrients). Stormwater flow preferentially removes fine particles from soil because of their lesser mass and tendency to remain suspended in flow once mobilized. Many of these pollutants form adsorption complexes that bind them to clay minerals, other fine particles, and organic matter. For this reason, many pollutants that associate with sediments are present in higher levels in the sediments mobilized by stormwater than in the parent soil (Novotny and Chesters 1989). The movement of these pollutants is therefore very similar to that described for sediment in *Section 2.2*.

Other pollutants dissolve in precipitation or disassociate (“desorb”) from sediment and move in a manner that mimics that of stormwater flows (e.g., nitrate, chloride). Some pollutants may switch between dissolved and particulate forms as conditions change. This may make control of these pollutants more difficult since standard practices for limiting sediment discharge may not significantly reduce these dissolved constituents.

Pollutants can also enter surface waters during dry weather due to excavation dewatering, groundwater seepage, construction equipment washout, landscape irrigation, and equipment operation in or near surface waters. Dewatering, the removal and discharge of water from excavated areas or work areas located within surface waters, is necessary at some construction sites to maintain dry work conditions. Water can collect in excavated areas due to precipitation and groundwater seepage (e.g., Horner et al. 1990, Montgomery County DEP 2009). Levels of dewatering activity and groundwater contamination vary widely among construction sites.

Excavation dewatering discharges can contain a variety of naturally occurring or human activity-derived pollutants. Any pollutant that can enter groundwater flow can be present in a dewatering discharge. Common groundwater contaminants include heavy metals, organic solvents and degreasers, pesticides and herbicides, and nitrates. Naturally occurring constituents, such as iron and manganese, can form extensive and unaesthetic deposits on surface water beds when groundwater under low reduction-oxidation (“redox”) conditions is exposed to ambient air.

3.1.2 National Scale and Cumulative Impact Concerns

The importance of the discharge of these other pollutants from construction sites is uncertain, but there are strong indications that the cumulative effects of construction activities is of national significance. Levels of pollutant discharge other than sediment and turbidity are highly variable among construction sites. EPA quantified construction site contributions of nitrogen and phosphorus, both common soil constituents, to surface waters. *Chapter 6* describes the methodology EPA used for this analysis as well as analysis results. Many other pollutants, however, may be less commonly found on construction sites. The current literature contains limited information on the frequency and level at which these pollutants appear at construction sites across the United States.

What literature is available indicates that individual construction sites typically do not discharge acutely toxic levels of metals and toxic organic compounds unless a material spill occurs or site soils and/or groundwater are contaminated with sufficient levels of toxic pollutants. However, discharges of pollutants below acutely toxic levels can nevertheless be of concern.

Many potential construction site pollutants are components of or adsorb to sediments (e.g., metals, toxic organic compounds, nutrients). Pollutants with this behavior can accumulate over time in surface water sediments to levels many times higher than those found in the adjacent water column. Pollutant levels can eventually become high enough to affect aquatic organisms (USFWS 1998). Construction is a widespread and frequent activity in the United States. The number of construction projects active each year (greater than 80,000) is much larger than the number of active facilities in many other industrial sectors (USEPA 2009b). The multiplier effect of such a large number of potential discharge points means that the construction sector’s cumulative pollutant contributions can be very high, even when contributions from individual sites are relatively low.

In addition, construction sites can discharge mixtures of pollutants. While no one pollutant may be present at a level sufficient to harm a surface water or its biota, the cumulative or synergistic effect of several mixture components may be sufficient to do so, particularly if other environmental stressors are present. Construction sites also discharge to many surface waters that are already impaired or in danger of impairment. Construction sites contribute to the already elevated cumulative pollutant loads in these waterbodies.

3.2 Construction Site Sources of Pollutants Other Than Sediment and Turbidity

3.2.1 Construction Materials and Equipment

An enormous variety of materials are used on construction sites. A number of these materials contain chemicals or produce debris or residues that can cause harm if they enter surface waters in sufficient quantities. These materials can contribute nutrients, metals, and toxic organic compounds to stormwater discharges.

A partial list of common building and equipment materials includes paint; sealants; grouts; solvents; adhesives; wood preservatives (e.g., creosote, copper azole, ammoniacal copper zinc arsenate); wood shavings, scraps, and sawdust; drywall; packaging material; detergents (e.g., trisodium phosphate); metal materials and debris; asphalt-based materials (e.g., roofing and paving materials); Portland cement; concrete; concrete truck washout; aggregate; dust suppressants (e.g., calcium chloride); blasting explosives; and construction equipment oil, fuel, lubricants, antifreeze, other fluids, and washwaters.

Pavement materials include asphalt, Portland cement, pavement sealers (e.g., methacrylate sealers, coal-tar based sealants) and additives (e.g., plasticizers, air entraining agents, water reducers, strength accelerating agents, corrosion inhibitors, colorants, pumping acids, rejuvenators, liquid and fibrous polymers, carbon black, sulfur). Many industrial wastes/by-products are used as aggregate or additives in asphalt and Portland cement pavement mixtures or as aggregate for unpaved roads. These materials include recycled asphalt and concrete pavement, shredded scrap tires, blast furnace slag, plastic waste, coal bottom ash, coal fly ash, steel slag, mine tailings and other wastes, municipal sludge, municipal solid waste incineration bottom ash, phosphogypsum, wood waste, petroleum refinery residuals, foundry sand, bricks, and asphalt shingles (National Research Council 2001; Eldin 2002).

Additional materials, both newly manufactured and industrial wastes/by-products, are used as soil amendments or fill on construction sites. Materials include imported soil and stone, lime (soil stabilization), cement (soil stabilization), crushed limestone, asphalt emulsions, and a variety of industrial by-products. Industrial by-products used as fill include coal combustion wastes (e.g., fly ash, bottom ash, boiler slag), lime kiln dust, gypsum, gypsum wallboard, foundry sand, sandblasting abrasives, cement kiln dust, wood ash, glass, and wastewater filter sand (Eldin 2002; USEPA 2005b; Iowa Waste Reduction Center 2009). Landscaping materials include topsoil, soil sterilants in treated topsoil, fertilizer, mulch, hydroseeding mixtures, lime, straw, and pesticides.

Pollutants from construction equipment and materials can be carried to surface waters in stormwater (or other) flow in solution or by erosion of debris and particulate matter (City of Vancouver 2007). Infiltration of construction site stormwater into soil can reduce transport of pollutants to surface water, though many pollutants adsorb to soil particles and will travel with them if they are subsequently eroded and transported to surface water. Improper operation and maintenance of construction equipment and insufficient storage and housekeeping practices (e.g., improper storage of oil, gasoline, and chemical products) can lead to additional leakage, spillage, or erosion of materials and their transport to surface waters. Spills at construction sites have been documented (Garton 1977 (diesel fuel); Ohio 1997f (construction debris); Kress 2007).

Some types of construction activity (e.g., stream channel realignment, culvert installation, bridge support emplacement) take place directly in surface waters. In addition, at some sites construction equipment must cross surface waters in order to access all parts of a site. These activities can provide easy access for contaminants from construction materials and equipment to surface waters, as well as cause direct disturbance of surface water sediments, physical structure, and organisms if precautions are not taken.

Sedimentation ponds can become sources of pollutants if they are allowed to become host to large amounts of algae, insect larvae, bacteria, and other organisms or if they are allowed to heat water to high temperatures before discharge.

The type and level of building, equipment, fill, soil amendment, and landscape material pollutants in stormwater discharges from a specific construction site depend on the choice of materials and construction practices used on the site. Many building, pavement, soil amendment, fill, and landscaping materials vary in composition based on source materials and manufacturing method. Pollutants present in a material from one source or manufacturer may not be present in the same category of material from another source or manufacturer (NRC 2001c). Not every member of every material category will contain constituents of concern.

3.2.2 Historic Site Contamination

Construction soil and groundwater disturbance can mobilize pollutants already present on a site due to prior human activity. A broad range of human activities and land uses can contaminate a site's soil and groundwater.

Industrial activities can directly deposit a wide range of pollutants in soil and groundwater, including metals, nutrients, and toxic organic compounds (Folkes et al. 2001; Seattle and King County Public Health 2001; Area Wide Soil Contamination Task Force 2003, Blanco et al. 2009).

Agricultural lands can contain elevated levels of pesticide residues, nutrients from fertilizers and manure (particularly phosphorus), and salts from irrigation waters (New Jersey DEP 1999; Area Wide Soil Contamination Task Force 2003; Renshaw et al. 2006; Robinson et al. 2007; Stone and Anderson 2009). Approximately 40 percent of current developed land was previously used for agricultural activities (NRC 2008).

In urban areas, construction often begins with demolition and removal of existing buildings and other structures. Materials in preserved wood (e.g., creosote, chromated copper-arsenate); paint chips and dust (including lead-based paints); brick and wood scraps; concrete debris; asbestos-based insulation and other materials; polychlorinated biphenyls (PCBs) from old electricity transformers; metal scrap; waste solvents; historic fuel spills; and fill material composed of coal ash and clinkers, brick and concrete demolition debris, and other industrial by-products may be present (Blanco et al. 2009). Pesticides have been widely used in urban areas, as well, and residues may persist in soil and other materials (Folkes et al. 2001). Erosion and leachate from these materials may elevate pH, metals, and toxic organic compounds levels in stormwater runoff.

Several contaminants can be deposited on the land through atmospheric deposition, including metals (e.g., copper, chromium, lead, mercury, and zinc), nutrients (nitrogen, phosphorus), and organic compounds (e.g., PAHs, PCBs, and pesticides). Sources of these contaminants include manufacturing facilities (e.g., smelters), power plants, motor vehicles, and wind-borne dusts (Folkes et al. 2001, NRC 2008). Atmospheric deposition from these sources is more likely to occur near urbanized areas and downwind of sources. However, atmospheric deposition can be very widespread and may affect areas that are fundamentally undeveloped (e.g., undisturbed forest). Over time, atmospherically deposited pollutants may accumulate to significant levels (Seattle and King County Public Health 2001, Area Wide Soil Contamination Task Force 2003). For example, lead emissions from several decades of leaded fuel use in the United States elevated lead levels in many urban soils.

The frequency and magnitude at which soil contamination is present nationwide is unknown. A task force sponsored by the state of Washington estimated that up to 676,550 acres in the state may contain soil contaminated with low to moderate levels of lead and/or arsenic (Area Wide Soil Contamination Task Force 2003). The state of New Jersey, which convened a similar task force, estimated that soil on up to 5 percent of the state's land was contaminated with lead and arsenic residues from the historical use of lead arsenate pesticides (New Jersey DEP 1999). At the severe end of the contamination scale, approximately 300 sites on the Superfund National Priorities List have been identified as having some level of soil contamination. At this time, EPA has determined that the soil contamination is severe enough at approximately 150 of the sites to require cleanup under the Superfund program (USEPA 2004c).

The type and level of historical pollutants at a specific construction site depend on the nature of the past activities and pollutant release and varies widely among sites. Not all sites contain constituents of concern from historic contamination.

3.2.3 Natural Site Constituents

Naturally occurring soil, rock, and groundwater composition and constituents vary widely among sites. Soil is a complex mixture of inorganic and organic materials deriving from weathered rock, biological activity, precipitation, and atmospheric deposition. Impacts from suspended solids, turbidity, and sedimentation from eroded soil are discussed in *Chapter 2*. Soil constituents also commonly include nitrogen, phosphorus, partially and fully decomposed organic matter, bacteria, fungi, insects, metals (e.g., iron, aluminum, magnesium, manganese), and other inorganic materials (e.g., compounds containing silicon, potassium, calcium, sodium, sulfur, inorganic carbon, and chlorides). Some constituents, such as zinc, copper, arsenic, chromium, nickel, barium, and lead, can be natural soil components but can be highly uneven in their distribution. Type and level of soil and rock constituents at specific construction sites depend on the nature of the soil and rock present on the construction sites. For example, Hawaiian soils have naturally higher levels of copper, zinc, and aluminum (Wong 2005).

Rock and soil constituents are natural components of sediments entering aquatic ecosystems. At elevated levels, however, they can become pollutants. Construction activity modifies natural erosion processes by pulverizing and exposing formerly sequestered rock and soil materials to erosion and leaching by precipitation, making transport of their constituents to surface waters much more likely.

Some sites also have natural groundwater seeps that may be further exposed or captured in site drainage by construction activity. These seeps, particularly those with naturally acidic waters or highly sodic waters, may contain elevated levels of certain constituents, including metals. Kalainesan (2007) documented elevated levels of dissolved solids and metals in a construction site groundwater seep (aluminum, manganese, iron, magnesium, and sulfate).

Elevated levels of pollutants in stormwater and receiving surface waters due to natural rock, soil, and groundwater constituents at construction sites has been documented in the literature (Huckabee et al.

1975; Burton et al. 1976, Chisolm and Downs 1978; Extence 1978; Tan and Thirumurthi 1978; Daniel et al. 1979; Shields and Sanders 1986; Kucken et al. 1994; Kalainesan 2007; USEPA 2009b).

Naturally occurring organisms such as bacteria and fungi can also function as pathogen pollutants. For example, *Aspergillus sydowii*, a common terrestrial fungus, can enter marine environments through soil erosion and discharge to surface waters and is a widespread sea fan pathogen in the Caribbean (Lafferty et al. 2004).

3.2.4 Altered Stormwater Discharge

An additional impact from construction activity is that vegetation removal, soil compaction, modified surface topography, and installation of impervious surfaces (e.g., roads, parking lots, and roofs) can increase the volume and intensity of water runoff discharging from a construction site during precipitation events but decrease a surface water's dry weather baseflow.

The increased flow can cause changes in the hydrograph of a receiving stream, causing it to exhibit more frequent and larger peak flows, followed by periods of reduced base flow and possibly desiccation. The stream may incise or widen to accommodate periodic elevated flow volumes (Wheeler et al. 2003). The changes in streambed morphology can increase sediment erosion, destabilize stream banks, elevate sediment and turbidity levels in downstream waters, and cause long-term changes in aquatic and riparian habitat. Elevated flows can also damage aquatic communities by scouring and washing downstream aquatic plants, algae, invertebrates, and other organisms. These changes can degrade or destroy previously existing aquatic communities.

Groundwater recharge is reduced by increased intensity and volume of stormwater runoff and reduced soil infiltration. Reduced groundwater recharge results in less water available for surface water dry weather baseflow. Reductions in dry weather baseflow can cause extended dry periods in streams and stress aquatic organisms.

The type and level of runoff from a specific construction site depends on the nature of the construction site, though many sites, particularly those that have been cleared of vegetation, have elevated surface runoff. Surface runoff changes are predictable through the use of standard hydrologic models (e.g., Soil Conservation Service TR-55, USDA 1986).

Line and White (2007) documented similarly elevated levels of stormwater runoff during all stages of a residential construction project in North Carolina. The authors postulated that soil compaction during the clearing and grading stages of the project were sufficient to mimic the impervious surfaces in place during later stages of the project. The authors also documented an increase in peak discharge rate and elimination of stream baseflow in the affected area. Selbig and Bannerman (2008) found that a stormwater basin serving a community under construction using Low Impact Development (LID) techniques discharged more water than the original undeveloped landscape but significantly less than a stormwater basin serving a neighborhood developed with conventional techniques. Clausen (2007) found that while runoff was elevated from a traditional construction site, it actually declined at a site using LID practices. Other studies documenting increases in surface runoff include Yorke and Davis (1972), Selbig et al. (2004), and Montgomery County DEP (2009). Burton et al. (1976) and Chen et al. (2009) did not detect a change in site runoff levels.

Construction sites can also concentrate previously diffuse surface runoff with ditches, pipes, and other structures. When a high volume of runoff enters a surface water, particularly at too high a velocity, it can erode the receiving water's banks and bed if not properly diffused. This erosion can modify local waterbody morphology, transport sediment, and elevate turbidity levels in downstream waters.

3.3 Other Pollutants from Construction Activity

This section provides additional information on specific pollutants potentially found in construction site discharges. The discussion describes potential sources of other pollutants, their transport behavior, and their potential to impact the aquatic environment and resources.

3.3.1 Nitrogen and Phosphorus

Phosphorus and nitrogen compounds are natural components of most aquatic ecosystems. At excessive levels, however, they become pollutants. Nitrogen and phosphorus are present at some level in nearly all construction site soils. Information on levels of nitrogen and phosphorus compounds typically found in soils and sediments is provided in *Section 6.4.1.* Several studies have documented discharge of nutrients from construction sites believed to derive from natural soil constituents (Burton et al. 1976; Tan and Thirumurthi 1978; Daniel et al. 1979; Shields and Sanders 1986).

Nutrients can also derive from construction materials and historic contamination. While some level of nitrogen and phosphorus is naturally present in most soils, the presence of nutrients from other sources varies widely. Roofing materials treated with phosphate washes and binders during preparation for use may release phosphorus (NRC 2008). Phosphorus from blasting explosive residue on a construction site has been documented (USEPA 2009b).

Fertilizers are also frequently used during the surface stabilization phase of construction to encourage vegetation growth. They can be applied directly or as a component of a hydroseeding mixture or other product. Organic materials such as mulch, compost, and straw are also used to encourage vegetation and contain elevated nutrient levels relative to site soils. When insufficiently managed, they can contribute nutrients to construction site discharges (Glanville et al. 2004). Faucette et al. (2007) documented elevated nitrogen and soluble phosphorus levels in construction site stormwater runoff from areas treated with straw blankets containing mineral fertilizer for erosion control. Hydroseeding operations, in which seed, fertilizers, lime and, in some cases, tackifiers, are applied to soil in a one-step operation are more likely to discharge nutrients than conventional seed-bed preparation operations during which fertilizers and lime are tilled into the soil. Nutrient discharges from fertilizer use on construction sites have been documented in several studies (Taylor and Roff 1986; Horner et al. 1990; Faucette et al. 2007; Clausen 2007; Kalainesan 2007; Faucette et al. 2008; Chen et al. 2009).

Decades of application of phosphorus-containing fertilizers and human, animal, and industrial waste to the land surface has significantly elevated phosphorus levels in many surface soil horizons in developed areas of the United States (Brady and Weil 1999). Land recently used as cropland may have elevated soil phosphorus levels from many years of fertilizer application. This phosphorus can be mobilized through erosion (Paul and Meyer 2001).

Other studies documenting the presence of nutrients in construction site stormwaters and/or their elevation in receiving waters due to construction activity include White (1976), Barton (1977), Barrett et al. (1995a), Harbor et al. (1995), Fossati et al. (2001), Kayhanian et al. (2001), Wong (2005), and Selbig and Bannerman (2008). Some studies monitored for and found no elevation in nutrient levels downstream of construction sites (Peterson and Nyquist 1972; Extence 1978; Young and Mackie 1991; Cleveland and Fashokun 2006).

Phosphorus and nitrogen can be found in both particulate and soluble forms on construction sites. Soluble nitrogen and phosphorus compounds are the typical form found in fertilizers so as to be immediately available to vegetation. Relatively low levels of soluble nitrogen compounds and very low levels of soluble phosphorus compounds also occur in natural terrestrial soils. Soluble nitrogen compounds leach more readily through soil than phosphorus compounds. Soluble forms of nitrogen and phosphorus can dissolve in and be transported with stormwater flows.

Most nitrogen in terrestrial soils is found in particulate organic form (Brady and Weil 1999). Phosphorus is typically found in particulate organic matter or bound to sediment. Nutrients form adsorption complexes with clay minerals and organic matter. Phosphorus readily binds to sediment, even when initially introduced to soil in soluble form such as fertilizer. Once bound to soil, it does not leach readily. Because phosphorus has such a strong association with sediment, discharges of total phosphorus tend to increase and decrease with total suspended solid discharges from sites (Selbig and Bannerman 2008). Discharges of organic particulate nitrogen also vary closely with suspended sediment discharges (Daniel et al. 1979).

Some studies have documented strong associations between turbidity or sediment discharges and nutrient discharges from construction sites, suggesting that discharged nutrients were associated with particulate matter eroded from the construction sites (Daniel et al. 1979; Shields and Sanders 1986). Daniel et al. (1979) found 99 percent of total phosphorus discharges and 90 percent of total nitrogen discharges from three residential construction sites to be associated with sediment. Because of stormwater's tendency to transport finer particles from eroded soils and nutrients' tendency to be associated with fine sediments and organic matter, eroded sediments are frequently enriched in nutrient content relative to the parent soils (Novotny and Chesters 1989). Once in a waterbody, nutrients can, under certain conditions, detach from sediments and enter the water column (Bilotta and Brazier 2008).

3.3.1.1 Nitrogen, Phosphorus, and Aquatic Resource Impacts

While nutrients are necessary for the primary production (organic matter derived from photosynthetic activity) that forms the base of many aquatic food chains, they can become pollutants at elevated levels. Eutrophication is a process by which excessive nutrient levels increase the growth rates of primary producers in aquatic systems, particularly algae, which can drastically alter a waterbody's ecological balance and impair human use of the waterbody (Wetzel 2000). Gradual eutrophication of some types of surface waters is a natural process, but nutrient levels elevated by human activity can drastically change the rate at which the process takes place (Goldman and Horne 1994).

Both nitrogen and phosphorus must be available for eutrophication to proceed. Freshwater system eutrophication is typically limited by phosphorus availability, and marine systems are typically limited by nitrogen availability. Because many undisturbed stream phosphorus levels are naturally low, even small increases can impact water quality. Nitrogen can also be the primary nutrient limiting freshwater primary production in some regions of the United States, particularly portions of the Northeast and the Pacific Northwest with granitic or basaltic geology (USEPA 2006d) and in other waters on a seasonal basis.

Elevated nutrient levels can change the quantity and quality of algal communities in both marine and freshwater systems. High concentrations of algal biomass, commonly referred to as algal "blooms," accumulate in the water column faster than natural processes (e.g., settling, zooplankton grazing, benthic organism filtering, senescence) can remove them.

Algal blooms can significantly reduce surface water transparency and therefore reduce the light available for submerged aquatic vegetation (SAV) photosynthesis. Elevated nutrient levels in the water column also encourage the growth of algae on the surface of submerged vegetation, which further decreases light availability. Light attenuation reduces photosynthesis levels and the maximum depth at which SAVs can survive. Deterioration in water quality due to eutrophication has been a major factor in the decline of SAV in many ecosystems (e.g., Staver et al. 1996). A number of estuaries have experienced loss of high quality SAV beds (e.g., eelgrass, turtlegrass) because of reduced light availability due to eutrophication from elevated nitrogen loadings. Impacts to aquatic vegetation subsequently decrease food and habitat availability for other aquatic organisms.

Excessive nutrients in shallow, wadeable streams can lead to dense mats of algal growth, particularly filamentous algae (e.g., *Cladophora* spp.) on bottom substrates. Such growths alter the habitat quality of the substrate and lead to predictable changes in aquatic macroinvertebrate populations. The proportion of pollutant-tolerant species in the community increases. Proportions and abundances of various feeding guilds shift, changing food quality and availability for fish and other organisms. Elevated nutrient levels, in combination with elevated levels of fine-grained sedimentation, can foster the growth of aquatic macrophyte beds, particularly in stream pools (King and Ball 1964; Taylor and Roff 1986).

Decomposition of excessive algal biomass reduces dissolved oxygen levels in surface waters, lessening the amount available to other aquatic organisms. This can impair organisms' functioning and health and possibly kill them. In surface waters stratified by temperature and/or salinity gradients, the settling of large amounts of organic matter can create low oxygen or anoxic conditions in bottom waters and allow only a few pollution-tolerant species to persist in the benthic community. Large diurnal swings in surface water oxygen content have also been observed in some waters. Supersaturation of dissolved oxygen takes place during daylight hours when photosynthesis takes place. At night, oxygen levels fall dramatically due to plant respiration. Such shifts have been shown to cause many summer fish kill events.

Coldwater fish species (e.g., salmonids) that depend on cool, well-oxygenated bottom waters as a refuge during summer are particularly vulnerable to eutrophication impacts and may be extirpated from affected systems. Low levels of dissolved oxygen can affect amphibians by reducing respiratory efficiency, metabolic energy, reproductive rate, and ultimately survival (USFWS 2005).

Eutrophication can also lead to shifts in fish community composition due to modifications of trophic feeding relationships. For example, the proportion of large, piscivorous fish species that rely on sight to catch their prey may decline in a surface water as water clarity falls. Filter-feeding planktivores (e.g., gizzard shad) may greatly increase in abundance. If eutrophication conditions are extreme, a fish community can be reduced to pollution-tolerant bottom feeders (e.g., carp).

Eutrophic conditions can also foster the growth of certain types of algal blooms with toxic effects on aquatic organisms and human beings. Brown or red tides are intense algal growths that can release neurotoxins into the water column, harming some forms of aquatic life. Toxic effects on aquatic organisms can cause population declines and localized extinctions (Burkholder 1998).

These types of blooms usually require beach closings and temporary bans on seafood harvesting because consuming shellfish from affected waters can cause shellfish poisoning in humans (Burkholder 1998). Human exposure to marine algal neurotoxins has been associated with a range of neurobehavioral abnormalities and is an emerging area of study (Friedman and Levin 2005). Illnesses most frequently linked to neurophysiological disturbance are Amnesic Shellfish Poisoning, Ciguatera Fish Poisoning, and Possible Estuarine Associated Syndrome, which is associated with exposure to substances from the dinoflagellate *Pfiesteria piscicida* through the food chain.

In freshwater systems, toxins from blue-green algal (Cyanophyta) species (e.g., *Microcystis*, *Cylindrospermum*) can be of concern. Blue-green algae toxins are classified according to mode of action and include hepatotoxins (e.g., microcystins), neurotoxins (e.g., anatoxins), skin irritants, and others (WHO 2003). Increased levels of organic material from algal blooms can also increase concentrations of chlorination disinfection by-products, including haloacetic acids and trihalomethanes, in treated drinking water. Severe morbidity and mortality in domestic animals due to toxin-contaminated drinking water has been documented (WHO 2003). Microcystins, in particular, have been associated with acute liver damage and possibly liver cancer in laboratory animals.

Some algal blooms may not be directly harmful to humans but may still impair human use of aquatic resources. Increased levels of organic material associated with algal blooms can clog drinking water intakes and cause unpleasant tastes and odors in drinking water (Boyd 1990) (see *Chapter 9*). Excessive algae growths can reduce the attractiveness and viability of surface waters for a variety of recreational activities including swimming, boating, fishing, wildlife viewing, and other outings. Excessive growths can also clog screens on water intakes for irrigation, industrial use, and drinking water.

Sedimentation levels also increase over the long term in eutrophic waterbodies, particularly lakes, reservoirs, and other impoundments with quieter flow conditions. Sediment cores obtained from studies of former lake environments have documented large increases in annual sediment deposits after the onset of eutrophication (Goldman and Horne 1994). These elevated sedimentation levels, acting in concert with sediment deposition from construction activities and streambed erosion due to land use change, can decrease depth and volume of surface waters over time. *Chapter 8* discusses EPA's analysis of economic impacts associated with loss of reservoir capacity.

Algal growth also increases surface water turbidity, the various aquatic life and human resource impacts of which are discussed in *Sections 2.3 and 2.4.*

The nitrogen compound ammonia can function as a nutrient, the impacts of which are discussed above. Ammonia is also toxic to many aquatic organisms at relatively low levels under certain water conditions. Ammonia toxicity is primarily attributable to the unionized form (NH_3) as opposed to the ammonium ion (NH_4+) form. Ammonia toxicity typically increases with pH such that above pH levels of 9, the most toxic form (unionized ammonia) is the predominant fraction. Temperature also influences ammonia toxicity. Low dissolved oxygen levels found in the bottom waters of eutrophic waterbodies can increase the potential for ammonia toxicity, particularly when a surface water is stratified. The low redox conditions found in these waters favor the anaerobic microbial conversion of nitrite or nitrate to ammonia.

3.3.1.2 Nitrogen and Phosphorus Criteria and Surface Water Impairment

As noted in *Section 2.6*, Section 305(b) of the Clean Water Act (CWA) requires states, territories, and other jurisdictions of the United States to submit reports to EPA on the quality of their surface waters every two years If a waterbody fails to meet any one of its designated uses, CWA Section 303(d) requires a state or other entity to list the waterbody as "impaired" and not meeting its designated uses. Excessive nutrients (phosphorus, nitrogen) and its associated biological response (i.e., chlorophyll a, DO levels, transparency) are a common reason for a waterbody failing to meet its designated uses.

Appropriate levels of nitrogen and phosphorus vary among surface waters. Nutrient levels associated with eutrophication vary among regions with the country due to differences in geology, climate, and soil types. EPA has published a series of technical guidance documents that provide methods for setting nutrient water quality criteria for (1) lakes and reservoirs (USEPA 2000c); (2) rivers and streams (USEPA 2000d); (3) estuaries and coastal marine waters (USEPA 2001); and (4) wetlands (USEPA 2008c). EPA has used this guidance to develop tables of recommended nutrient criteria for lakes/reservoirs and streams/rivers for several ecoregions across the country (USEPA 2009d). States, tribes, and other entities have started to develop their own numeric nutrient criteria, using EPA recommendations as a starting point.

The Assessment TMDL Tracking and Implementation System (ATTAINS) provides information on water quality conditions reported by the states to EPA under Sections 305(b) and 303(d) of the Clean Water Act. The information available in ATTAINS is updated as data are processed and are used to generate the biennial National Water Quality Inventory Report to Congress. This information reflects only the status of those waters that have been assessed.

Table 3-1 presents information on surface waters with nutrient-related impairments (algal growth, ammonia, noxious aquatic plants, nutrients, and organic enrichment/oxygen depletion) by EPA Region as reported by the states in ATTAINS. *Appendix A* provides information on the state water report year for which data were available for populating the table below as of September 17, 2009.

It should be noted that individual waters may be impaired by more than one pollutant. Although states tend to target their monitoring efforts to those surface waters they believe to be impaired, the total area of impaired surface waters due to nutrients is probably underestimated due to the low percentage of surface waters that were assessed. As of September 17, 2009, states had assessed only 26 percent of the nation's reach miles, 42 percent of its lake acres, and 20 percent of its bay and estuary square miles.

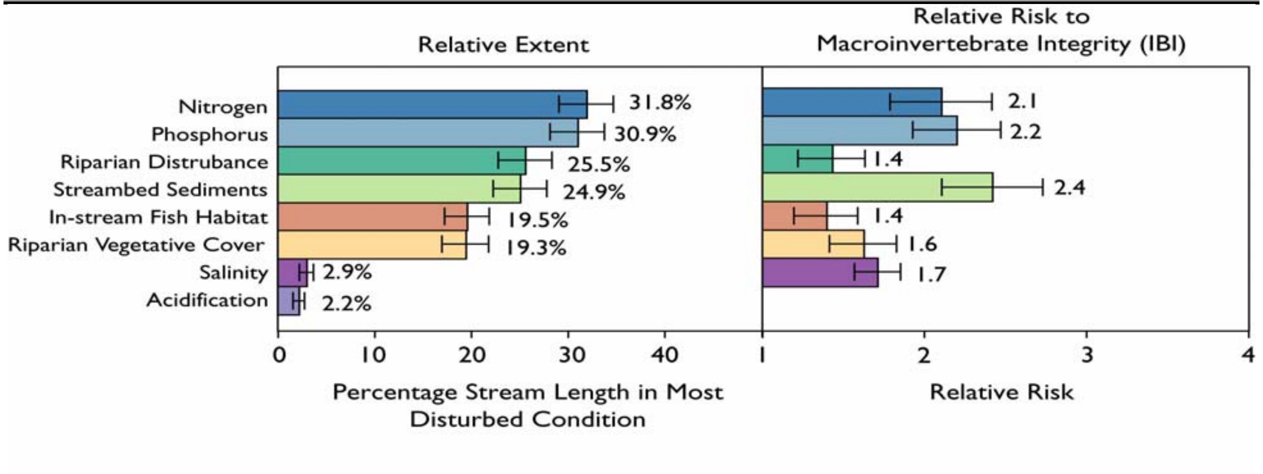
Table 3-1: Nutrient-Related Impairment in 305(b)-Assessed Waters, by EPA Region

EPA Region	Stream/River (miles)	Lake/Pond/Reservoir (acres)	Bay/Estuary (sq. miles)
1	2,267	229,728	759
2	10,485	219,203	591
3	8,009	61,179	4,039
4	11,522	262,978	22
5	72,350	1,244,029	
6	13,162	742,606	724
7	22,369	487,346	
8	11,298	995,262	
9	20,816	804,315	111
10	31,722	506,696	
Nation	204,000	5,553,342	6,246

Source: EPA ATTAINS database (USEPA 2009a) as of 9/17/09.

Water quality in the United States has also been assessed through a series of national, probability-based surveys known as the National Aquatic Resource Surveys. These surveys use randomized sampling designs, core indicators, and consistent monitoring methods and laboratory protocols to provide statistically defensible assessments of water quality at the national scale. The *Wadeable Streams Assessment* (USEPA 2006d) is a statistical survey of the smaller perennial streams and rivers that, according to the report, comprise 90 percent of all perennial reach (i.e., stream and river) miles in the United States. Elevated nutrient and sedimentation levels are the top stressors of streams. According to the survey, 32 percent of streams have "poor" nitrogen conditions, and 21 percent have "fair" conditions relative to reference streams. Also, 31 percent have "poor" phosphorus conditions, and 16 percent have "fair" conditions relative to reference streams. The survey also examined the association between stressors and biological condition and found that high levels of nitrogen or phosphorus more than double the risk for poor biological condition (see *Figure 3-1*).

Figure 3-1: Extent of Stressors and Their Relative Risk to the Biological Condition of the Nation's Streams



Source: USEPA (2006d).

Several threatened and endangered (T&E) species are vulnerable to eutrophication from nutrient pollution. One study postulated that 25 percent of all current freshwater T&E species are adversely impacted by eutrophication (Richter et al. 1997). Mollusks are frequently vulnerable to eutrophication impacts. Examples include several species of freshwater snails and mussels (USFWS 2000; Biber 2002; and Kozlowski and Vallelian 2009). Fish species and aquatic plants are also vulnerable. Threatened fish species include the Waccamaw silverside (*Menida extensa*), paleback darter (*Etheostoma pallididorsum*), and an arctic grayling subspecies (*Thymallus arcticus montanus*) found in Montana and Wyoming (NatureServe Explorer 2009).

To quantitatively evaluate the water quality benefits from reducing nitrogen and phosphorus loadings from construction sites, EPA developed an approach that relies on the empirical relationship between in-stream sediment and nitrogen and phosphorus levels. A full discussion of this approach is provided in *Chapter 6*.

3.3.2 Organic Compounds and Materials

A wide variety of organic compounds and materials can be discharged from construction sites. Organic compounds are present on construction sites as natural soil constituents, components of a variety of building and equipment materials, and historic soil contamination. The nature and frequency of these materials' occurrences on and discharge from construction sites has not been well characterized in the literature. Given the proprietary nature of the composition of many commercially available products and the enormous variety of possible compounds in these materials, research in this area can be challenging.

Pavement materials such as asphalt and petroleum-based asphalt pavement rejuvenators contain a variety of organic compounds. There are many concrete additives whose environmental effects have not been thoroughly characterized (e.g., air entraining agents, water reducers, strength accelerating agents, and pumping acids) (NRC 2001c). Benzothiazole has been detected in leachate from shredded scrap tires commonly used as pavement additives and is a likely toxin (Eldin 2002). N, 4-dimethylbenzamine has been detected in methacrylate bridge deck sealer and is also highly toxic (NRC 2001c).

PCB compounds have been banned for use in manufacturing in the United States due to their carcinogenicity, persistence in the environment, and bioaccumulation potential. Because they were widely used historically and are very stable in the environment, their residues are frequently found in soils of areas where they have been manufactured or used, including many industrial and urban sites. PCBs bind well to sediments and are often transported with them.

Organic compounds can have toxic effects on aquatic organisms and/or influence surface water oxygen levels when present at sufficient levels. Discharges of organic compounds and materials of all types show a strong correlation to suspended sediment discharges (Barrett et al. 1995b). Oil and grease, other petroleum hydrocarbons, PAHs, and many organic pesticides adsorb to sediment and travel with it as it erodes. As sediments accumulate in receiving waters, concentrations of associated organic pollutants can increase to levels above those found in the adjacent water column. Over time these sediment levels can become high enough to cause adverse impacts to organisms living in, interacting with, or connected through the food chain to those sediments.

Information on some of the better characterized pollutants known to be of concern, including high organic content material, petroleum hydrocarbons, PAHs, and organic pesticides, is provided below.

3.3.2.1 Organic Matter and Dissolved Oxygen

Organic matter consists of biodegradable carbon-based material. Organic material is a natural constituent of many construction site soils. A common construction site practice is to stockpile topsoil, roots, and other vegetative debris from site clearing on-site for use in final site landscaping. If not protected from erosion, these stockpiles can contribute organic matter, sediment, turbidity, and adsorbed pollutants to surface waters. Developers may also import topsoil for use during landscaping. Wood mulch, compost, straw, organic geotextiles (i.e., biodegradable structural sheets used to reduce slope erosion), and other organic materials used for soil erosion control and landscaping can increase organic material discharges if improperly maintained and allowed to erode (Glanville et al. 2004). Taylor and Roff (1986) documented extensive erosion of straw mulch from a construction site, sufficient to clog culverts downstream of the site.

Elevated loadings of organic material can increase levels of oxygen-demanding substances (e.g., as measured by biological oxygen demand (BOD) and chemical oxygen demand (COD)) in receiving waters. Microbes aerobically break down the organic compounds. Elevated BOD and COD levels can lower dissolved oxygen levels in surface water, leading to several of the impacts associated with nutrient-derived oxygen depletion discussed in *Section 3.3.1*.

Excessive algal growth has been documented in sedimentation basin waters at some construction sites (Kalainesan 2007; USEPA 2009b). These blooms can deplete oxygen levels diurnally and elevate organic matter content in water discharged from these basins.

Some studies have documented a decline in the organic matter content of surface water sediment due to construction activity (Barton 1977; Young and Mackie 1991). This effect is derived from the erosion of largely inorganic soil particles from construction sites and the high levels of those sediments entering the surface water. Some aquatic organisms depend on organic matter in surface water sediments as a food source. Elevated levels of sediment containing low levels of organic matter can reduce the food value of this material for these organisms.

3.3.2.2 Petroleum Hydrocarbons

Petroleum hydrocarbons include a variety of crude and refined oil products, including gasoline, diesel fuel, and lubricating oils. These products contain a diverse array of organic compounds, many poorly characterized in terms of environmental behavior and impact. Petroleum hydrocarbons are often monitored in water as a group, either as total petroleum hydrocarbons or as oil and grease.

Petroleum-derived aliphatic hydrocarbons on construction sites typically derive from human activity, either current or historic. Many historic industrial sites have some level of petroleum hydrocarbon contamination in site soils and, at some sites, groundwater as well. Improper operation, fueling, and maintenance of construction equipment, and poor housekeeping practices (e.g., improper storage of oil, diesel fuel, and gasoline products) can lead to leakage or spillage of materials containing petroleum hydrocarbons (Garton 1977). Accidental rupturing of petroleum transport lines can also occur during construction (USFWS 2005, citing two incidents during utility line trench construction in Texas).

Asphalt-based products (pavement material, asphalt shingles) are commonly used on construction sites and contain a variety of organic compounds including n-alkanes, carboxylic acids such as n-alkanoic acids, benzoic acids, thiaarenes, and PAHs (see below). Kayhanian et al. (2001) documented oil and grease in construction site stormwaters.

Petroleum hydrocarbons tend to adsorb to soil and sediment particles and travel with them as they erode and settle in surface waters. Petroleum hydrocarbons discharged directly to water spread quickly to cover water surfaces. Microbial activity in surface waters eventually degrades the more structurally simple components of petroleum materials (e.g., 40–80 percent of a crude oil) (Hoffman et al. 1995).

Petroleum hydrocarbons can affect aquatic organisms through direct toxicity, smothering, and changes in water quality. Water quality impacts can include light reduction, pH alteration, dissolved oxygen reduction, and dissolution of some product components in the water column. Oil, grease, fuel, and other petroleum hydrocarbons typically contain toxic and carcinogenic constituents. Organisms may be impacted directly or through impacts to food sources (USFWS 2005, Hoffman et al. 1995).

Fish can be exposed to petroleum hydrocarbons through contact with dissolved fractions in the water column, particulate fractions and contaminated sediments and ingestion of contaminated food and water. Floating egg masses can come into contact with surface hydrocarbon layers (Malins and Hodgins 1981). Garton (1977) documented a diesel spill at a construction site that resulted in a fish kill. These dynamics affect other aquatic organisms, as well. Waterfowl can be affected by external oiling, ingestion, egg oiling, and habitat changes.

3.3.2.3 Polycyclic Aromatic Hydrocarbons

PAHs are a large and diverse class of organic compounds deriving from both natural and anthropogenic sources. Natural sources include forest fires, volcanic emissions, and oil seeps. Anthropogenic sources include thermal combustion by industrial, municipal, power plant, vehicular, and household entities. PAHs from these sources can be deposited from the atmosphere onto soil over wide areas. They can accumulate in soils, particularly in urbanized areas, and subsequently erode after ground disturbance.

PAHs are also found in construction materials. Coal tar-based sealcoat for driveway and parking lot surfaces contains very high levels of PAHs (Mahler et al. 2005; Van Metre et al. 2006) and has been found to contaminate stormwater and sediment in downstream surface waters (Science Daily 2009). It has been inferred that these sealcoat products are contributing to PAH contamination of many U.S. lakes.

PAHs are also found in oil, asphalt, and tar products used for pavement and roofing projects, some foundry sands, creosote wood preservatives, and shredded scrap tires used as a pavement additive. The PAHs in asphalt include sulfur-containing PAHs, such as dibenzothiophene, which are more water soluble and bioaccumulative than some other PAHs (NRC 2008). Several studies have noted road construction sites as possible contributors of PAHs to surface waters (Ohio EPA 1998d, 1998e, 1999b).

A number of commonly used solvents (e.g., toluene, trichloroethane, and dichloroethane) can transport PAHs and have frequently contaminated soil and groundwater at industrial and urban sites. Historic industrial sites may contain PAHs from other sources as well, such as disposal of coal tar and combustion by-products. Historic fill materials on industrial and urban sites may also contain PAHs.

In general, PAHs have low solubility in water, high melting and boiling points, and low vapor pressure (Hoffman et al. 1995). PAHs are persistent in the environment and do not break down easily in water. They also have bioaccumulation potential (USEPA 2008d). In the aquatic environment, most PAHs associate with soil and sediment particles rather than partitioning to the water column. Their adsorption to sediment increases with increasing organic content of the sediment. PAHs may accumulate in sediments to concentrations much higher than those measured in the water column.

PAHs can adversely affect mammals (including human beings), birds, fish, amphibians, invertebrates, and plants. PAHs at low concentrations can stimulate or inhibit growth of aquatic bacteria and algae. At higher concentrations, however, they interfere with cell division and photosynthesis (Eisler 1987). Effects on aquatic invertebrates include inhibited reproduction, delayed emergence, sediment avoidance, and mortality. Egg and larval stages are more susceptible. Effects on amphibians and reptiles include impaired reproduction, reduced growth and development, tumors, and cancer. Effects on fish include fin erosion, liver abnormalities, cataracts, and immune system impairments (Eisler 1987; Van Veld et al. 1990; Mahler et al. 2005; USFWS 2005). PAHs from application of coal tar-based pavement sealcoat have been documented as the cause of a fish kill (Maryland DEP 2000).

PAHs can transfer to humans through consumption of fish from surface waters contaminated with sufficient levels of PAHs (USEPA 2004b). PAHs have been found to cause birth defects and liver and blood problems in some animals (USEPA 2008d). PAHs are a known carcinogen in many animals and are suspected to be carcinogenic in human beings as well (USEPA 2008d).

3.3.2.4 Organic Pesticides

A large number of pesticides are organic compounds. Organic pesticides have been and continue to be used extensively on both agricultural and urban lands (e.g., building foundation and lawn pest control). Some pesticides, including many organochlorine pesticides used historically but since banned in the United States, create residues that are able to persist in soils for long periods of time. These residues can erode when soils are disturbed. Pesticide residues on construction sites can derive both from activities associated with construction (e.g., landscaping and vegetation removal activity) as well as from historical pesticide applications on urban and agricultural lands that are now being redeveloped. Chlorpyrifos and diazinon have been detected in construction site stormwaters in California (Kayhanian et al. 2001).

Organochlorine pesticides that strongly adsorb to clay and organic matter and can be transported by erosion and overland runoff include 1,1,1-trichloro-2,2-bis(p-chlorophenyl)ethylene DDT, aldrin, mirex, kepone, dieldrin, endosulfan, toxaphene, lindane, heptachlor, chlordane, and difocol (Alberta ARD 2009). Except for endosulfan, use of these pesticides was generally banned in the United States because of their toxicity, persistence in the environment, and/or potential to bioaccumulate. For example, even though DDT usage peaked and began to decline in the 1960s and was banned in 1972 in the United States, DDT and its breakdown products - DDD (1,1-dichloro-2,2-bis(p-chlorophenyl)ethane) and DDE (1,1-dichloro-2,2-bis(p-chlorophenyl)ethylene) persists widely in many agricultural soils and surface water sediments in current and former agricultural watersheds (New Jersey DEP 1999).

Persistence, bioaccumulation, and toxicity properties vary among organic pesticides. Some organic pesticides in current use are generally believed to degrade more quickly once applied in the environment and have fewer toxic effects.

Due to their hydrophobic nature, pesticides tend to adsorb to sediments and organic materials and, therefore, tend to erode, travel, and settle with sediment particles in surface waters. Similar to other organic compounds, pesticides are often found at low or undetectable levels in the water column, but can accumulate to problematic levels in surface water sediments, particularly fine-grained organic materials. Biomagnification of pesticides occurs as higher trophic level organisms, such as fish-eating birds and mammals, ingest pesticides in prey. Ingestion can also occur during feeding activities that involve sediment contact (e.g., probing of sediment with bill, grooming).

Once in a surface water, pesticides can remain associated with sediment or can detach and enter the water column given appropriate conditions (Bilotta and Brazier 2008). Aquatic organisms come into contact with pesticide residues during contact with or ingestion of contaminated water, sediment, and prey. Herbicides can remain active even at low concentration levels and damage aquatic vegetation (USFWS 1998). Several studies have documented morphological, developmental, and biochemical alterations in a variety of amphibians exposed to atrazine (USFWS 2005).

Most human sediment-related exposure to contaminants is through indirect routes in which pollutants transfer from sediments to the water column or to aquatic organisms. A number of fish consumption advisories or fishing bans are due to chlordane, DDT, and its metabolites (DDD and DDE), all of which are commonly found in sediments.

3.3.3 Metals

Metals derive from a variety of sources on construction sites. Iron, aluminum, and manganese are common constituents of natural soils. Other metals can also occur naturally, but their distribution is highly variable. Elevated concentrations of these other metals are more commonly associated with human activity. A variety of metals can be a concern in the aquatic environment. These include aluminum, antimony, arsenic, cadmium, chromium, copper, iron, lead, manganese, mercury, nickel, selenium, silver, and zinc.

Construction equipment and materials can be sources of metals. A wide variety of metal materials are used for roofing, pipes, and structural frameworks and supports. Galvanized metal (e.g., for pipes, fence supports, and roofing) can elevate zinc levels in construction site stormwater runoff (Horner et al. 1990; NRC 2008). Cement and concrete products, wastes, and equipment washwaters can have elevated metal levels (e.g., of chromium) due to industrial wastes incorporated during the manufacturing process (USEPA 2006a). High levels of aluminum and iron have also been documented in cement wastes (NRC 2008). Preserved and waterproofed wood products can leach high levels of copper (NRC 2008). One study attributed elevated levels of construction site metal discharge (chromium) to pressure-treated wood products (Ohio EPA 1997d).

A variety of materials used for road and highway construction contain metals, including pressure-treated wood, coal fly ash, phosphogypsum, and asphalt mixes containing municipal incinerator ash, shredded scrap tires, foundry sand, and recycled shingles. In laboratory studies, these materials have been found to leach heavy metals at levels toxic to water fleas (*Daphnia magna*) and green algae (*Selenastrum capricornutum*). Leached metals included arsenic, copper, zinc, lead, aluminum, mercury, and vanadium (USEPA 1999; NRC 2001c; Eldin 2002). Although adsorption to soil particles ameliorates the metal leachates' toxic effects (NRC 2001), these soil particles may subsequently erode from construction sites and enter surface waters. Loose particles of road and highway construction materials can also erode and enter surface waters.

Landscaping materials can contain metals, as well. Some fertilizers contain elevated levels of metals due to industrial wastes incorporated during their manufacture. Glanville et al. (2004) documented elevated levels of several metals in composted biosolids, yard waste, and bioindustrial waste relative to topsoil. Compost can be used to reduce total erosion (and therefore total loadings of metals) from construction sites. However, if compost products contain elevated metal levels and are allowed to erode, they can contribute metals to construction site discharges. Metals content varies widely among composted products and is heavily dependent on the source materials used.

Historic contamination can also be a source of metals. Lead is a common soil contaminant near roads and in urbanized areas due to its use in lead-based paint and as a gasoline additive for many years. Soil in fruit orchards active prior to 1950 are frequently contaminated with high levels of lead and arsenic due to widespread use of lead arsenate pesticides during the late 19th and early 20th centuries (New Jersey DEP 1999). These pesticides were also widely used for golf course and lawn care (Folkes et al. 2001). Soils on lands used to grow cotton prior to 1950 often contain high levels of arsenic due to use of arsenic acids to defoliate cotton prior to harvest. Lead and arsenic pesticide residues do not decay over time and remain in surface soil layers until a disturbance, such as construction activity, mobilizes them through erosion.

Metals have been documented in construction site stormwaters and at elevated levels in downstream receiving waters due to construction activity. Documented metals include aluminum (Huckabee et al. 1975), cadmium (Kayhanian et al. 2001), chromium (Kayhanian et al. 2001), copper (Horner et al. 1990; Barrett et al. 1995a; Kayhanian et al. 2001; Wong 2005; Clausen 2007), iron (Extence 1978; Shields and Sanders 1986; Barrett et al. 1995; Chen et al. 2009), lanthanum (Huckabee et al. 1975), lead (Shields and Sanders 1986; Horner et al. 1990; Kayhanian et al. 2001; Clausen 2007), manganese (Huckabee et al. 1975; Shields and Sanders 1986), nickel (Kayhanian et al. 2001), samarium (Huckabee et al. 1975), silver (Kayhanian et al. 2001), and zinc (Huckabee et al. 1975; Shields and Sanders 1986; Horner et al. 1990; Barrett et al. 1995a; Kayhanian et al. 2001; Clausen 2007). Some studies attributed metals to natural constituents in site soils or groundwater (Huckabee et al. 1975; Extence 1978; Shields and Sanders 1986; Yew and Makowski 1989; Barrett et al. 1995a). In many studies, the source of the metals was not identified. The presence and level of metals is highly variable among site discharges. Some studies examined but detected no elevation in metals levels in receiving waters downstream of construction sites (Peterson and Nyquist 1972).

Heavy metals can form complexes with clay minerals and organic matter in soil or can be a component of particulate matter in soil. Organic matter, in particular, has a high capacity to bind with metals. Sediments with higher levels of organic matter typically have higher concentrations of zinc, lead, chromium, copper, mercury, and cadmium (Paul and Meyer 2001).

Metals also tend to associate with finer sized sediment particles. Because stormwater tends to transport finer particles from sites, the sediment in stormwater tends to be enriched in metals relative to the parent soil. Novotny and Chesters (1989) discussed metals that can be present at elevated levels in sediment relative to the original soil, including lead, copper, zinc, aluminum, iron, chromium, and nickel.

Metals also show a strong correlation with suspended solids discharges. One study of construction site discharges noted strong correlations between total suspended solids (TSS) and particulate copper, zinc, and chromium discharge levels, suggesting that the metals travel with sediment as it erodes (Kayhanian et al. 2001). Barrett et al. (1995a) found strong correlations between iron and TSS discharge concentrations downstream of a construction site. Barrett et al. (1995b) stated that lead and iron are the metals most strongly associated with particulate matter, followed by copper and cadmium.

Metals tend to adsorb to sediment in the aquatic environment. Because metals do not biodegrade, they can accumulate on waterbody beds as they settle from the water column with sediments. Due to this accumulation in bottom sediments over time, metals levels may often be below levels of concern or detection in the water column but reach levels high enough to have adverse impacts on organisms living in or near bottom sediments. For example, lead concentrations tend to be highest in benthic organisms, perhaps due to feeding among bottom sediments (USFWS 1998). Metals can also desorb from sediments and enter the water column under appropriate conditions (e.g., anaerobic conditions) (Bilotta and Brazier 2008).

During dredging or flooding, sediment containing metals may be reintroduced into the water column because of the disturbance. Metals attached to suspended sediment can be transported downstream, potentially contaminating areas far from their point of origin. Disturbance of contaminated sediments can also release metal contaminants into water drawn from surface waters for drinking water supply. When present at sufficiently high levels, metals can cause dredged sediment to be classified as toxic, making its disposal more costly.

A variety of organisms in streams in urbanized areas have exhibited elevated metal concentrations, including algae, mollusks, arthropods, and annelids. Organisms are directly exposed to both dissolved metals in the water column and metals associated with ingested sediments and organic matter. The potential for toxicity depends on the bioavailability of the metal, the means and length of organism exposure, and the life stage exposed. The bioavailability of the metal depends on the nature of the metal discharge and the aquatic environment.

Acute and chronic ambient water quality criteria have been established for most heavy metals and contain adjustment factors to reflect dissolved and particulate metal forms and water column hardness (USEPA 2007d). A recent examination of the water quality criteria for copper by EPA uses the Biotic Ligand Model (BLM) and incorporates data on receiving water levels of dissolved organic carbon, pH, major cations and anions, alkalinity, and temperature. The presence of other types of pollutants or materials can be important as some act additively, synergistically, or antagonistically with certain metals. For example, the bioavailability of cationic metals in sediments decreases with increasing organic content and the presence of binding sulfide compounds (USEPA 2005b).

While trace amounts of certain metals are necessary for normal biological functions in many aquatic species, higher concentrations can be acutely or chronically toxic and adversely affect behavior, growth, development, metabolism, reproductive success, and survival in aquatic species, waterfowl, and mammals. Exposure to metals has been shown to reduce photosynthetic efficiency and algal colonization, lowering primary production rates and diminishing food available to other organisms. Elevated metal levels can inhibit plant growth and can have adverse effects on the health of benthic organisms (Masterson and Bannerman 1994). Metals can be absorbed through skin, gills, intestines, and other organs. Early organism life stages and organisms with longer exposure durations are more vulnerable (USFWS 2005). Some metals are able to progress and bioaccumulate through the food chain, imposing heavier metal exposures on organisms higher in the tropic structure. Impacts from elevated metals levels can reduce organism abundance and alter community structure.

Iron and manganese are more common and less toxic to aquatic organisms than heavy metals. They are highly sensitive to redox conditions and are more soluble under low dissolved oxygen conditions. High levels of dissolved iron and manganese can be found seasonally in anaerobic bottom waters of stratified lakes. High levels can also be found in low-oxygen groundwaters. In solution, these metals can create highly colored water. Low-oxygen groundwaters containing high levels of iron and/or manganese may be discharged from construction sites due to grading, excavation, and dewatering activities. If these groundwaters are discharged to surface waters, aeration and rapid oxidation can deposit large amounts of red iron oxides or black manganese oxides on surface water substrates. These deposits can coat stream substrate and aquatic vegetation, fill interstitial spaces, and create turbid, highly colored water. Chisholm and Downs (1978) discuss the discharge of such groundwater from a construction site in West Virginia.

3.3.4 Dissolved Inorganic Ions

A number of dissolved inorganic ions can discharge from construction sites, including calcium, chloride, sodium, potassium, magnesium, sulfate, and bicarbonate. These ions are naturally occurring components of many soils and, in some cases, of precipitation. Levels vary geographically. Elevated levels relative to freshwater surface waters can be found in some groundwater. Construction site activity may increase discharge of ions to surface waters through soil disturbance and erosion, groundwater disturbance, use of dust suppressants (e.g., calcium chloride), fertilizers (which may contain chloride, potassium, or other inorganic ions), lime (calcium carbonate), cement, concrete, and other building materials. High levels of calcium, magnesium, and sodium have been documented in cement wastes (NRC 2008).

Several studies have documented the presence of dissolved inorganic ions in construction site stormwaters and at elevated levels in downstream receiving waters due to construction activity. Ions that have been documented include calcium (Huckabee et al. 1975; Tan and Thirumurthi 1978; Nodvin et al. 1986; Shields and Sanders 1986), bicarbonate (Tan and Thirumurthi 1978; Foassati et al. 2001), chloride (Tan and Thirumurthi 1978; Shields and Sanders 1986; Chen et al. 2009), magnesium (Huckabee et al. 1975; Nodvin et al. 1986; Shields and Sanders 1986), potassium (Nodvin et al. 1986), sodium (Tan and Thirumurthi 1978; Nodvin et al. 1986), and sulfate (Huckabee et al. 1975; Tan and Thirumurthi 1978; Shields and Sanders 1986; Chen et al. 2009). Some studies monitored but did not detect changes in dissolved inorganic ion levels in receiving waters downstream of construction sites (Burton et al. 1976; Cramer and Hopkins 1982; Hedrick et al. 2006).

Shields and Sanders (1986) and Chen et al. (2009) identified soil erosion as the source of ions. Chisholm and Downs (1978) identified construction site use of a calcium chloride dust suppressant as the source. Huckabee et al. (1975) identified acid rock drainage as the source. Nodvin et al. (1986) documented elevated levels of magnesium, sodium, and potassium in a stream due to leaching of material from concrete freshly poured for bridge footers.

Dissolved inorganic ions generally exhibit low toxicity to aquatic organisms. However, ions can influence several surface water attributes including salinity and pH (discussed below). They can also influence the toxicity of co-discharged metals. For example, large concentrations of calcium in surface waters can decrease toxicity of several metals, including aluminum (Fredman 1995). In wetlands and anaerobic sediments, however, microbial processes can exchange calcium ions with mercury and facilitate the release of mercury into these systems.

Elevated or widely fluctuating chloride levels can impair aquatic organism growth, reproduction, and survival (New Hampshire DES 2007). Some aquatic plants are sensitive to salinity changes (e.g., some studies of the genus *Potamogeton* have shown it to be very sensitive to water column salinity and mineral or ionic profile changes (USFWS 1998)). Irrigation water that contains dissolved salts or pollutants can harm crops and damage soil quality (Clark et al. 1985).

Depending on whether fish are fresh or salt water species, they have been reported to tolerate chloride levels of 400 to 30,000 mg/L. EPA has set an acute freshwater criterion for chloride at 680 mg/L and a chronic criterion at 230 mg/L (USEPA 1988). Chloride toxicity increases when it co-occurs with other ions such as potassium or magnesium. Dissolved inorganic ions can also affect organism osmoregulation. Some sensitive species (e.g., mayflies) have been lost from surface waters due to elevated ion levels (NRC 2008).

3.3.5 pH Level

Construction sites use materials that can alter surface water pH levels, including large quantities of cement and concrete. Wastes from concrete truck and other equipment washouts have a high pH (12 pH units) and can alter surface water pH if discharged at sufficient levels (Canadian DFO 2009). Stormwater can also become elevated in pH if it comes into contact with freshly placed concrete. Sand-cement bags may be used as temporary headwalls. Nodvin et al. (1986) documented elevated pH levels in a stream in Nevada downstream of a bridge construction site. pH levels increased from 6.83 to a maximum of 11.22. The authors postulated that leaching of calcium oxide hydration products from the concrete poured for the bridge footers was the source.

Some materials used on construction sites for soil stabilization and fill are also high in pH. These include lime and cement materials, cement kiln dust, lime kiln dust, and crushed limestone. Lime can also be added to soils as a fertilizer.

Natural site soil and rock constituents, such as limestone or sulfide-containing rock and soil, can also alter surface water pH if discharged at sufficient levels. Huckabee et al. (1975) documented acidification of a small stream in Tennessee due to iron sulfides in rock used for roadbed fill. Upstream of the fill, stream pH was 6.5 to 7.0. Downstream of the fill for several miles, stream pH was 4.5 to 5.9. Acidification took place shortly after completion of the project and continued to persist for more than 30 years. Yew and Makowski (1989) also documented stream acidification due to use of pyritic shales in road construction. EPA (USEPA 2009b) described a highway construction site in Washington where disturbance of sedimentary shale rock raised pH levels as high as 9.6 in stormwater and dewatered groundwater. Chen et al. (2009) documented significant increases in sulfate levels downstream of a construction site. The authors believed the sulfate derived from construction site erosion and contributed to increased stream acidity after project completion. Shields and Sanders (1986) also attributed pH changes in downstream receiving waters to soil constituents.

Change in pH levels in receiving waters was highly variable among sites. In addition to the studies described above, several studies monitored but detected no change in receiving water pH levels due to construction activity (Peterson and Nyquist 1972; Barton 1977; Extence 1978; Lenat et al. 1981; Cline et al. 1982; Cramer and Hopkins 1982; Taylor and Roff 1986; Foasatti et al. 2001; Hedrick et al. 2006).

Aquatic communities have low species richness at pH levels above or below 6 to 8.5 units (Kalff 2002). Atmospheric acid deposition has been documented to eliminate species as pH decreases and to extirpate fish from lakes once long-term levels fall below 5 pH units. At high (alkaline) pH levels, LC_{50} (lethal concentration) values for salmonids fall in the 9–10 pH unit range. Few fish survive even short-term exposures to pH levels greater than 11 (Alabaster and Lloyd 1980). Some aquatic plants are also sensitive to pH changes (e.g., some studies of the genus *Potamogeton* have shown it to be very sensitive to water column pH changes) (USFWS 1998). Low species richness at high or low pH levels can be attributable, in some cases, to other factors working in concert with altered pH levels. These factors can include increased metal toxicity (e.g., aluminum), high salt levels, and high water temperature.

3.3.6 Pathogens

Pathogens may be present on construction sites due to the use of manure products for fertilization and landscaping and improper disposal of on-site sanitary wastes. Pathogens are transported by attachment to sediment particles. The transport and fate of bacterial indicator species *Escherichia coli* and *Salmonella* spp. have been shown to be highly influenced by their relationship with flocculated suspended sediment and bed sediments in a river (Droppo et al. 2009). Bacteria counts were consistently higher within sediments than within the water column. Bed sediments were found to represent a possible reservoir of pathogens for subsequent remobilization and transport

Fungi can also function as pathogens. *Aspergillus sydowii*, a common terrestrial fungus, can enter marine environments through upstream erosion. This fungus has emerged as a widespread sea fan pathogen in the Caribbean. Another fungus dispersed in eroded soil is known to cause a disease called “valley fever” in California sea otters (Lafferty et al. 2004).

4 Summary of Literature on Construction Site Discharges to Surface Waters

A large number of publicly available studies issued in peer-reviewed, government, and other publications document pollutant discharges from construction sites to surface waters. EPA reviewed many of these studies and refers to the information they contain throughout this document. This chapter provides an overview of the studies EPA reviewed (*Section 4.1*) as well as a brief summary of each individual study (*Section 4.2*). *Section 4.3* presents state reports of construction site discharges.

4.1 Overview of Impacts in Literature

Studies documenting the results of research on construction site discharges have been published for many years. The earliest paper summarized in this chapter was published in 1959, and the most recent papers were published in 2009. Research characterizing the nature and magnitude of construction site discharges is ongoing.

Available studies cover a range of construction types, geographic locations, and surface water types. Studies of the impact of highway construction are more numerous than studies of impacts from other types of construction. Studies of impacts to streams and small rivers are more numerous than studies of impacts to other types of surface waters. Impacts to marine systems are poorly characterized. EPA was unable to locate an explanation for these aspects of the literature. One possible explanation for the frequency of stream ecosystems in the literature is the high frequency with which they are affected by construction site discharges and the ease with which they can be divided into “altered” and “unaltered” zones (i.e., controls) for study of construction site discharge effects.

Most of the studies EPA chose for review examined surface water impacts downstream of construction sites. EPA also reviewed a smaller number of studies characterizing the nature of construction site stormwaters prior to their discharge to receiving waters.

The literature indicates that construction sites can affect a number of stormwater and receiving water parameters, including physical, chemical, and biological parameters. Most studies, however, characterized only a subset of all potentially affected parameters, perhaps due to resource limitations. Suspended sediment and turbidity were the most commonly monitored parameters. Basic water quality parameters (pH, dissolved oxygen, some major dissolved ions, biological oxygen demand (BOD)), nutrients, metals, and toxic organic compounds were also monitored. Nutrients were monitored in a wide variety of forms, though only a subset of possible forms was studied in most cases. Likewise, when metals were monitored, only a subset of possible metals was typically chosen for observation. Specific organic compounds were rarely monitored. Summaries of study findings for specific pollutants are provided in *Chapters 2* and *3*.

Studies varied in whether they measured in-stream pollutant concentrations, total pollutant loads, or both. Measurement of pollutant concentrations was more common than measurement of pollutant loads. Each measurement provides a different type of information. Measurement of pollutant concentration is useful for determining certain acute and chronic effects associated with individual discharge events (e.g., surface water appearance, organism toxicity). Total load measurements provide information on the cumulative effects of a discharge source and can be particularly important for pollutants such as sediment, metals, and certain toxic and persistent organic compounds that can settle and persist in surface waters for several or more years.

A complication with the measurement of pollutant concentrations in construction site stormwater is that they can be diluted by the greater volume of stormwater runoff generated by many construction sites. An observer relying solely on pollutant concentration data to determine the effects of construction activity on stormwater discharges may incorrectly conclude that a construction site has no impact because concentration measurements do not reflect the cumulative impact of the total volume of runoff from the site. For example, Clausen (2007) observed that copper, lead, and zinc concentrations in stormwater runoff were unchanged by construction activity at one site, even though the total load of these pollutants discharged from the site increased.

Physical stream condition was discussed in a number of studies, particularly when sedimentation impacts were noted. Observed impacts included increased substrate embeddedness, complete burial of substrate beneath a sediment layer, elimination of pools and other stream microhabitats, changes in channel morphology, and loss of stream baseflow.

Biological parameters were monitored less frequently than water quality parameters. Macroinvertebrates, particularly benthic macroinvertebrates, were monitored most frequently, followed by monitoring of fish. Only a few studies examined impacts to primary producers or animals other than macroinvertebrates or fish (e.g., amphibians, mollusks, reptiles). Impacts to organisms were noted in most studies in which they were monitored. Impacts included reduction of abundance, diversity, and biomass. Increased drift and avoidance behavior, change in community composition, and mortality were also documented.

Other study limitations included limits on the spatial and temporal extent of monitoring. Few studies were designed to characterize the full spatial extent of impacts from construction sites' discharges. Most studies examined conditions within surface waters less than a mile downstream of construction sites. Several studies documented surface water impacts several miles downstream of construction sites (King and Ball 1964; Wolman and Shick 1967; Vice et al. 1969; Huckabee et al. 1975; Hainly 1980; Nodvin et al. 1986; Ohio EPA 1997f). One study documented impacts as far as 56 miles downstream (Fossati et al. 2001).

Many studies were also limited in the length of time they were able to monitor, and most were not designed to characterize the full temporal extent of impacts. Most monitoring took place during active construction. A number of studies included monitoring during pre-construction and post-construction periods, as well. Elevated suspended sediment and turbidity concentrations in downstream surface waters were generally noted to decline within a year or less of construction activity cessation and site surface stabilization with sufficient vegetation.

A number of studies documented longer-term impacts from construction site discharges. These impacts included elevated sedimentation of surface water substrates (Guy 1963; Chisholm and Downs 1978; Tsui and McCart 1981; Cline et al. 1982; Taylor and Roff; Stout and Coburn 1989; Barrett et al. 1995a; Reid and Anderson 1999). Elevated sedimentation levels in areas immediately downstream of construction sites were typically documented as requiring one to several years to clear. Monitoring of sedimentation of downstream surface water substrates as construction sediments migrated downstream was typically not included in the studies. Lee et al. (2009) noted that many years may be required after completion of construction in a watershed for construction-associated sediments to fully migrate downstream.

Longer-term effects to macroinvertebrate and fish populations were also documented (Peterson and Nyquist 1972; Huckabee et al. 1975; Reed 1977; Chisholm and Downs 1978; Tsui and McCart 1981; Cline et al. 1982; Taylor and Roff 1986; Reid and Anderson 1999). Macroinvertebrate population improvements were usually noted to occur as excess sedimentation was flushed from surface water substrates, and fish population improvements were usually noted as substrate and macroinvertebrate conditions improved. Recovery was often documented to require more than a year following construction completion, with some studies noting recovery as still incomplete several years after construction cessation (Taylor and Roff 1986; Montgomery County DEP 2009). In one case, damage was still noted 30 years after construction cessation because of stream acidification from iron sulfide rocks used as roadbed fill (Kucken et al. 1994).

Implementation of sediment and erosion controls varied widely among sites described in the studies, though a large number of sites, including many in older studies, utilized them to some degree. Requirements for use of sediment and erosion controls on construction sites have gradually become more stringent over time. Older studies tend to document discharges from sites with lower levels of sediment erosion and control. However, although older studies often do not reflect current requirements, they do provide valuable information on the nature of pollutant discharges from construction sites and their effects on surface waters. In addition, current practices vary among states and localities and are not always successfully implemented (particularly during large precipitation events) or maintained over time. Because current requirements have reduced, rather than eliminated, construction site discharges, information from older studies continues to be relevant for full evaluation of potential impacts.

Construction site discharges are dependent on site conditions (e.g., slope, vegetative cover, soil type), rainfall, nature of construction activity, presence and effectiveness of sediment and erosion abatement practices, and other factors. Given the large number of construction sites active every year (greater than 80,000) and limitations in resources available for construction site discharge and impact study and documentation, the collection of studies summarized here only partially describes the nature of construction site discharges taking place each year in the United States.

To quantify current national water quality impacts from construction site sediment discharges and how they would decrease under various regulatory options, EPA used several methodologies, including Spatially Referenced Regressions on Watershed Attributes (SPARROW) models, to estimate surface water sediment, nitrogen, and phosphorus levels deriving from construction activity. These methodologies and the results of the analyses are described in more detail in *Chapter 6* of this document.

Table 4-1 summarizes information from studies EPA reviewed that studied the effect of construction site activity on physical, chemical, and/or biological parameters of site stormwaters or receiving waters. Specific information on suspended sediment concentration, suspended sediment yield, and turbidity levels are provided in *Table 4-2*, *Table 4-3*, and *Table 4-4*.

"Altered Water Characteristics" as listed in *Table 4-1* are water condition alterations identified by a study as being due to construction activity. "Unaltered Water Characteristics" are water conditions identified by a study as unchanged by construction activity. In some studies, conditions listed under "Unaltered Water Characteristics" changed, but the cause was identified by the study as something other than construction activity.

Table 4-1: Studies Documenting Construction Site Discharges to Surface Waters and Their Impacts

Study	Location	Construction Type	Monitoring Points	Sediment & Erosion Controls	Altered Water Characteristics	Unaltered Water Characteristics
Barrett et al. (1995a)	Texas	Highway	Stream	Rock Berms Silt Fence	BOD5[1] Chemical Oxygen Demand (COD) Copper Fecal Coliform[1] Fecal Strep Iron Nitrate Sedimentation Total Coliform Total Organic Carbon[1] TSS Turbidity Volatile Suspended Solids Zinc	Cadmium Chromium Lead
Barton et al. (1972)	Utah	Highway	Stream	Unknown	Macroinvertebrates	
Barton (1977)	Ontario, Canada	Highway	Stream Ponds	Sedimentation Basin Straw Mulch Turfing	Benthic Macroinvertebrates Fish Nitrate Sedimentation Sediment Organic Content Suspended Sediment	Ammonia Dissolved Oxygen Hardness Nitrite pH Phosphate
Burton et al. (1976)	Florida	Highway	Stream Lake	Earthen Berms Hay Bales Mulching Plastic Sheeting Sedimentation Basins Seeding Sod Placement Visqueen Slope Drains	Orthophosphate [load] Phosphorus, Total Dissolved [load] Silicon, Dissolved [load] Suspended Sediment [conc. + load] Turbidity	Dissolved Solids [load] Stormwater Discharge
Carline et al. (2003)	Pennsylvania	Highway – Park Avenue	Stream	Rock Dam Sedimentation Basin Silt Fence Vegetative Buffer	TSS	Benthic Macroinvertebrates Substrate Composition Trout Redds
		Highway – Rock Road	Stream	Sedimentation Basin Silt Fence	TSS	Benthic Macroinvertebrates Substrate Composition Trout Redds

Table 4-1: Studies Documenting Construction Site Discharges to Surface Waters and Their Impacts

Study	Location	Construction Type	Monitoring Points	Sediment & Erosion Controls	Altered Water Characteristics	Unaltered Water Characteristics
Chen et al. (2009)	West Virginia	Highway	Stream	Grass seeding Mulch Sedimentation Basin Silt Fence	Acidity Alkalinity (possible) Calcium (possible) Chloride Conductivity (possible) Iron Macroinvertebrates Nitrate Sulfate TSS Turbidity	Ammonia Phosphate Temperature Water Discharge
Chisholm & Downs (1978)	West Virginia	Highway	Stream	Seeding	Dissolved Solids Macroinvertebrates Sedimentation Stream Substrate	
Clausen (2007)	Connecticut	Residential - Traditional	Storm Sewer	Unspecified Practices	Ammonia [yield] Copper [yield] Lead [yield] Nitrate + Nitrite [yield] Runoff Volume TKN [conc.][1] TKN [yield] Total Phosphorus [conc.] Total Phosphorus [yield] TSS [yield] Zinc [yield]	Ammonia [conc.] Copper [conc.] Lead [conc.] Nitrate + Nitrite [conc.] TSS [conc.] Zinc [conc.]
		Residential – Enhanced Best Management Practices (BMPs)	Ditch	Earthen Berm Hay Bales Silt Fence Topsoil stockpile cover	Ammonia [conc.] Copper [conc.] Lead [conc.] Nitrate + Nitrite [conc.] Runoff Volume[1] TKN [conc.] Total Phosphorus [conc.] Total Phosphorus [yield] TSS –[conc.] TSS –[yield] Zinc [yield][1]	Ammonia [yield] BOD [conc.] Copper [yield] Fecal Coliform [conc.] Lead [yield] Nitrogen [yield] TKN [yield] Zinc [conc.]
Cleveland & Fashokun (2006)	Texas	Highway	Ditch	Rock Filter Dam	TSS Phosphorus	Ammonia Nitrate Nitrite Turbidity

Table 4-1: Studies Documenting Construction Site Discharges to Surface Waters and Their Impacts						
Study	**Location**	**Construction Type**	**Monitoring Points**	**Sediment & Erosion Controls**	**Altered Water Characteristics**	**Unaltered Water Characteristics**
Cline et al. (1982)	Colorado	Highway	Stream	Unknown	Epilithon Macroinvertebrates Substrate composition TSS	Dissolved Oxygen pH Total Dissolved Solids
Cramer & Hopkins (1982)	Louisiana	Highway	Wetland	Unknown	Color Turbidity	Dissolved Oxygen pH Salinity
Downs & Appel 1986	West Virginia	Highway	Stream	Unknown	Suspended Sediment	
Duck (1985)	Scotland, Great Britain	Road	Stream Lake	Unknown	Sedimentation Sediment Load Suspended Sediment Turbidity	
Eckhardt et al. (1976)	Pennsylvania	Highway	Stream	Mulching Seeding	Suspended Sediment	
Embler & Fletcher (1983)	South Carolina	Unknown	Stream	Unknown	Suspended Sediment Turbidity	
Extence (1978)	Great Britain	Highway	Stream River	Unknown	Iron Macroinvertebrates Sedimentation Suspended Sediment	BOD Nitrate Orthophosphate pH
Fossati et al. (2001)	Bolivia	Highway	River	Unknown	Bicarbonates Macroinvertebrates Phosphates Sedimentation Suspended Sediment Turbidity	Calcium Chlorides Conductivity Magnesium Nitrates pH Potassium Sodium Sulfates Temperature
Garton (1977)	West Virginia	Highway	Spring	Unknown	Fish Petroleum Hydrocarbons Suspended Sediment	
Guy (1963)	Maryland	Residential	Stream	Unknown	Sedimentation Suspended Sediment	
Hainly (1980)	Pennsylvania	Highway	Stream	Unknown	Suspended Sediment	

Table 4-1: Studies Documenting Construction Site Discharges to Surface Waters and Their Impacts

Study	Location	Construction Type	Monitoring Points	Sediment & Erosion Controls	Altered Water Characteristics	Unaltered Water Characteristics
Hedrick et al. (2006)	Tennessee	Highway	Stream	Buffered Disposal of Acidic Rock Rock Check Dams Silt Fence	None	Calcium Conductivity Magnesium pH Suspended Sediment
Hedrick et al. (2007)	West Virginia	Highway	Streams	Silt Fence	Benthic Macroinvertebrates (short term) Sediment Yield	Benthic Macroinvertebrates (long term) Sediment Size Fractions
Helm (1978)	Pennsylvania	Highway	Stream	Unknown	Suspended Sediment	
Helsel (1985)	Ohio	Highway	Stream River	Benches Dams Dikes Excelsior Matting Hay Bales Jute Matting Mulching Rock-Lined Channels Sedimentation Basins Seeding	Suspended Sediment	
Huckabee et al. (1975)	North Carolina	Highway	Stream	Unknown	Aluminum Bicarbonate Calcium Fish Lanthanum Magnesium Manganese Periphyton pH Salamander Samarium Stream Substrate Sulfate Zinc	Arsenic Cadmium Chloride Cobalt Copper Iron Lead Mercury Phosphate Potassium Selenium Sodium
Hunt & Grow (2001)	Ohio	Unknown	Stream	None	Fish Habitat Quality Indices Sedimentation Substrate Embeddedness	
Keller (1962)	Maryland	Multiple	River	Unknown	Suspended Sediment	

Table 4-1: Studies Documenting Construction Site Discharges to Surface Waters and Their Impacts

Study	Location	Construction Type	Monitoring Points	Sediment & Erosion Controls	Altered Water Characteristics	Unaltered Water Characteristics
King & Ball (1964)	Michigan	Highway	River	Unknown	Fish Macroinvertebrates Primary Production Sedimentation Suspended Sediment Turbidity	
Lee et al. (2009)	Kansas	Commercial Highway Residential	Stream	Unknown	Sediment Yield	
Lenat et al. (1981)	North Carolina	Highway	Stream	Sedimentation Basin Silt Fence	Macroinvertebrates Substrate Composition Suspended Sediment	Dissolved Oxygen pH Temperature
Line (2009)	North Carolina	Highway	Streams Lake	Coir Baffles Flocculent Mulching Rock Dams Sediment Traps Sedimentation Basins Seeding Silt Fence Skimmer Outlet Slope Drains Turbidity Curtain	Suspended Sediment Yield Turbidity	
Line & White (2007)	North Carolina	Residential	Stream	Sediment Trap	Ammonia Nitrate Peak Runoff Runoff – Rainfall Ratio Stream Baseflow TKN Total Nitrogen Total Phosphorus TSS	
Lubliner & Golding (2005)	Washington	Commercial Residential Transportation Utility	Stream	Blanket Erosion Control Mulch Storm Drain Protection Sedimentation Basin Vegetation	Transparency TSS Turbidity	

Table 4-1: Studies Documenting Construction Site Discharges to Surface Waters and Their Impacts

Study	Location	Construction Type	Monitoring Points	Sediment & Erosion Controls	Altered Water Characteristics	Unaltered Water Characteristics
Montgomery County DEP (2009)	Maryland	Multiple	Streams	Baffles Basin Forebays Limit Disturbance Oversized + Dual Perforated Risers Sedimentation Basin – Skimmers Stabilization	Channel Aggradation Channel Sinuosity[1] Fish Macroinvertebrates Sedimentation Stream "Flashiness" Substrate Embeddedness TKN TSS	
Nodvin et al. (1986)	California	Bridge	Stream Lake	Unknown	Alkalinity Calcium Conductivity Magnesium pH Potassium Sodium Turbidity	
Ohio EPA (1997f)	Ohio	Golf Course Residential	Stream	Sedimentation Basin	Fish Macroinvertebrates Mollusks Sedimentation Substrate Embeddedness Turbidity	
Owens et al. (2000)	Wisconsin	Residential Commercial	Stormwater Discharge	Not included in study	Suspended Sediment	
Peterson & Nyquist (1972)	Alaska	Highway Bridge	Stream	Unknown	Conductivity Macroinvertebrates Turbidity	Alkalinity Ammonia Calcium Carbon Dioxide Dissolved Oxygen Iron Magnesium Nitrate Nitrite Orthophosphate pH Silica Sulfate Temperature

Table 4-1: Studies Documenting Construction Site Discharges to Surface Waters and Their Impacts

Study	Location	Construction Type	Monitoring Points	Sediment & Erosion Controls	Altered Water Characteristics	Unaltered Water Characteristics
Reed (1977)	Virginia	Highway	Streams	Hay Bales Mulching Rock Dam Sedimentation Basin Seeding	Fish Macroinvertebrates	
Reed (1980)	Pennsylvania	Highway	Streams	Mulching Rock Dams Sedimentation Basins Seeding	Suspended Sediment	
Reid & Anderson (1999)	Multiple Sites in North America	Pipeline	Streams Rivers	Unknown	Benthic Macroinvertebrates Channel Morphology Fish Sedimentation Substrate Composition Substrate Embeddedness Suspended Sediment Turbidity	
Selbig et al. (2004)	Wisconsin	Residential	Stream	Construction Phasing Deep Tilling Earthen Berms Inlet Protection Silt Fence Stockpile Seeding Stone Tracking Pads Straw Bales Vegetative Buffers	Runoff Volume Suspended Sediment Total Sediment	Fine Sediment Fish Habitat Quality Macroinvertebrates Stream Morphology Temperature
Selbig & Bannerman (2008)	Wisconsin	Residential	Stormwater Basin	Earthen Berms Erosion Fabric Rock Trenches Sediment Basins Seeding Silt Fence	Phosphorus - total Runoff Volume Total Solids TSS	

Table 4-1: Studies Documenting Construction Site Discharges to Surface Waters and Their Impacts

Study	Location	Construction Type	Monitoring Points	Sediment & Erosion Controls	Altered Water Characteristics	Unaltered Water Characteristics
Shields & Sanders (1986)	Mississippi	Waterway	Stream	Polymer Flocculant Sedimentation Basin Disposal Area Design	Multiple[2]	Arsenic – total [conc.] Cadmium – total [conc.] Chromium – total [conc.] Copper – total [conc.] Mercury – total [conc.] Nitrate – total [conc.] Nitrate + Nitrite [conc.] Organic Nitrogen [conc.[Phosphorus – dissolved [conc.] TKN [load] Zinc – total [conc.]
Stephens et al. (1996)	Utah	Highway	Stream Reservoir			
Tan & Thirumurthi (1978)	Nova Scotia, Canada	Highway	Lake	Diversion Channel Sedimentation Basin	Conductivity Nitrate + Nitrite Total Dissolve Solids Turbidity	Multiple[3]
Taylor & Roff (1986)	Ontario	Highway	Stream	Sedimentation Basin Straw Mulch Turfing	Fish Macroinvertebrates Macrophytes Nitrate Sedimentation Sediment Organic Content Suspended Sediment	Ammonia Dissolved Oxygen Hardness pH Phosphorus
Tsui & McCart (1981)	British Columbia, Canada	Pipeline	Stream	Unknown	Macroinvertebrates Sedimentation TSS Turbidity	
Vice et al. (1969)	Virginia	Highway	Stream	Unknown	Suspended Sediment Turbidity	
Walling & Gregory (1970)	Devon, England	Residential	Stream	Unknown	Suspended Sediment	
Ward & Appel (1988)	West Virginia	Highway	Stream	Unknown	Suspended Sediment	
Wark & Keller (1963)	Maryland, Virginia, and West Virginia	Multiple	Streams Rivers	Unknown	Suspended Sediment	

Table 4-1: Studies Documenting Construction Site Discharges to Surface Waters and Their Impacts

Study	Location	Construction Type	Monitoring Points	Sediment & Erosion Controls	Altered Water Characteristics	Unaltered Water Characteristics
Welsh & Ollivier (1998)	California	Highway	Streams	Unknown	Amphibians Fine sediment depth Substrate Embeddedness	
Werner (1983)	West Virginia	Highway	Spring	Unknown	Fish Suspended Sediment	
Whitney & Bailey (1959)	Montana	Highway	Stream	Unknown	Fish	
Wolman & Schick (1967)	Maryland	Commercial	Stream	Unknown	Sedimentation Stream Substrate	
Wong (2005)	Hawaii	Highway	Stream Reservoir	Unknown	Multiple[4]	Multiple[5]
Yew & Makowski (1989)	Tennessee-North Carolina Border	Highway	Stream	Unknown	Alkalinity Fish Metals pH	
Yorke & Herb (1978)	Maryland	All types	Streams	Unknown	Suspended Sediment [yield]	
Young & Mackie (1991)	Northwest Territories, Canada	Pipeline	Stream	Stream Stabilization Vegetative Buffer	Macroinvertebrates Sedimentation TSS	Dissolved Oxygen Hardness Nitrogen – dissolved pH Phosphorus –dissolved

[1] Parameter declined between baseline and construction estimates.

[2] Alkalinity, Ammonia- total [conc.], Biological Oxygen Demand [conc.], Calcium – dissolved [conc.], Carbonate [conc.], Chemical Oxygen Demand [conc.], Chloride [conc.], Color, Hardness, Iron – total [conc. AND daily loading], Lead-total [conc. AND daily loading], Magnesium – dissolved [conc.], Manganese- total [conc. AND daily loading], Nitrate + Nitrite [daily loading], Nitrite – total [conc.], pH, Phosphorus-total [conc.], Specific Conductance, Sulfate [conc.], Total Dissolved Solids [conc. AND daily loading], Total Kjeldahl Nitrogen (TKN) [conc.], Total Nitrogen [conc. AND daily loading], TSS [conc.], Turbidity, and Zinc-total [daily loading].

[3] Alkalinity, ammonia, color, coliforms, dissolved oxygen, hardness, iron, manganese, pH, potassium, silica, temperature, total organic carbon

[4] Copper (possible), Nitrite + Nitrate, Sediment Yield, Specific Conductance, Total Phosphorus, Total Suspended Solids, Turbidity.

[5] 2,4-D, 2,4,5-T, Aldrin, Aluminum, Arsenic, Barium, Beryllium, Cadmium, Chlordane, Chlorpyrifos, Chromium, Cobalt, DDD, DDE, DDT, DEF, Diazinon, Dichlorprop (2,4-DP), Dieldrin, Dissolved Oxygen, Disulfoton, Endosulfan-alpha, Endrin, Ethion, Fonofos, Heptachlor, Heptachlor epoxide, Iron, Lead, Lindane, Lithium, Malathion, Manganese, Mercury, Methyl-Parathion, Methoxychlor, Mirex, Molybdenum, Nickel, Parathion, Perthane, pH, Phorate, PCB, PCN, Selenium, Silver, Silvex (2,4,5-TP) Strontium, Temperature, Total Nitrogen, Toxaphene, Trithion, Vanadium, Zinc.

Table 4-2, *Table 4-3*, and *Table 4-4* summarize findings on suspended sediment concentrations, sediment yield, and turbidity levels, respectively, from the studies reviewed by EPA. Sediment in construction site stormwaters or receiving waters was measured either as a concentration (*Table 4-2*) or a total yield from the site or the surface water's watershed basin (*Table 4-3*). Suspended sediment concentrations are measured in milligrams per liter (mg/L). Sediment yield is total sediment load expressed as a mass of sediment normalized by the area of the contributing watershed. It represents the mass of sediment eroded from the land, channels, and mass wasting, minus the sediment that is redeposited prior to the point of measurement. Sediment yield is expressed in terms of total yield or in terms of an increase due to construction site discharge and is sometimes specified per unit area or period of time. Turbidity is expressed in terms of NTUs (*Table 4-4*).

For most studies, the data presented in *Table 4-2*, *Table 4-3*, and *Table 4-4* reflect the averaging of multiple samples rather than the comparison of single grab samples. The number of samples and the length of time over which they are averaged vary widely among studies.

Table 4-2: Summary of Suspended Sediment Concentrations from Selected Studies

Study	Location	Suspended Sediment Concentration (mg/L)	
		Unaffected Site	**Affected Site**
Barrett et al. (1995a)	Texas	13.9	79
Barton (1977)	Ontario, Canada	2.8	5-50.0[1]
Chisholm & Downs (1978)	West Virginia	—	3,051.2
Clausen (2007)	Connecticut – Enhanced site	4[2]	67
	Connecticut – Traditional site	No statistical difference.	
Cleveland & Fashokun (2006)	Texas	212.5	1410.3
Cline et al. (1982)	Colorado	16.4	10.8
	Colorado	2.2	18.9
	Colorado	3.8	91.0
	Colorado	2.6	121.6
Duck (1985)	Scotland	0.3-3.4	7.4-712.8
Embler & Fletcher (1983)	South Carolina	<30[3]	60-130[3]
Fossati et al. (2001)	Bolivia	4	5-2,574[4]
Hedrick et al. (2006)	Tennessee	No statistical difference.	
Helsel (1985)	Ohio	125.0	261.0
Lubliner & Golding (2005)	Washington	—	33.7
	Washington	—	131.2
	Washington	—	628.7
Owens et al. (2000)	Wisconsin – summer construction	100	15,000
	Wisconsin – winter construction	550	2,400

Table 4-2: Summary of Suspended Sediment Concentrations from Selected Studies

Study	Location	Suspended Sediment Concentration (mg/L) Unaffected Site	Affected Site
Reed (1980)	Pennsylvania – Site 2	7-8[5]	12-18[5]
		42-100[6]	146-243[6]
	Pennsylvania – Site 2A	3-5[5]	9-14[5]
		38-63[6]	100-209[6]
	Pennsylvania – Site 2B	11-17[5]	58-70[5]
		65-77[6]	309-408[6]
	Pennsylvania – Site 3	8-9[5]	16-30[5]
		36-57[6]	227-285[6]
Shields & Sanders (1986)	Mississippi	119	347
Taylor & Roff (1986)	Ontario, Canada	5-10	30-206
Tsui & McCart (1981)	British Columbia, Canada	0-7	0-10,660
Wolman & Schick (1967)	Maryland - Cockeysville	1,500	80,000
Wolman & Schick (1967)	Multiple sites in Baltimore, Maryland and District of Columbia region		3,000 to 150,000 +
Wong (2005)[7]	Hawaii – Site 16265700	5	10
	Hawaii – Site 16266500	3	2
	Hawaii – Site 16267500	3	12
	Hawaii – Site 16269500	2	5
	Hawaii – Site 16270900	5	2
	Hawaii – Site 16275000	3	5
Young & Mackie (1991)	Northwest Territories, Canada	<2	>300

[1] Average concentration varied with stage of construction activity.
[2] Modeled estimate.
[3] Peak values.
[4] Study sites extended up to 56 miles downstream of construction activity.
[5] Average daily-mean base-flow values.
[6] Average daily-mean storm event values.
[7] Wong (2005) measured concentration as a long-term geometric mean.

Studies vary in the degree to which construction activity increased suspended sediment yield for multiple reasons, including variability in the percentage of the watershed under study containing construction activity, nature of construction activity and associated land disturbance, precipitation intensity, site characteristics (e.g., soil type and slope), and sediment and erosion control practices utilized. A number of the yields in *Table 4-3* were calculated for watersheds containing other land uses besides construction (e.g., forest, mature development, pasture, cropland). This method of calculation lowers estimated sediment yields because it incorporates lower sediment yields associated with these other land uses.

Table 4-3: Summary of Sediment Yield Data from Selected Studies

Study	Location	Sediment Yield: Unaffected Site	Sediment Yield: Affected Site	Units
Burton et al. (1976)	Florida		Up to 67	tons/acre/year
Carline et al. (2003)	Pennsylvania – Rock Road		5	tons/acre/year
Clausen (2007)	Connecticut – Enhanced BMP site	0.002[1]	0.024	tons/acre/year
	Connecticut – Traditional BMP site	0.0008[1]	0.10	tons/acre/year
Daniel et al. (1979)	Wisconsin – Site G2	–	12.27	tons/acre/year
	Wisconsin – Site G3		7.58	tons/acre/year
	Wisconsin – Site G5		5.84	tons/acre/year
Downs and Appel (1986)	West Virginia	0.2	9.7	tons/acre/year
Duck (1985)	Scotland, Great Britain	0.03	3.8	tons/acre/year
Eckhardt (1976)	Pennsylvania		62-103	tons/acre/year
Guy (1963)	Maryland	–	39	tons/acre/year
Guy and Ferguson (1962)	Virginia	–	2	tons/acre/year
Helsel (1985)	Ohio	0.7-1.2	15-25	tons/acre/year
Line (2009)	North Carolina – Tilly Up	0.01	7.3	tons/acre/year
	North Carolina – Tilly Down	0.07	3.50	tons/acre/year
	North Carolina – Ellery Up	0.04	2.02	tons/acre/year
	North Carolina – Ellery Down	0.20	1.35	tons/acre/year
	North Carolina – King's Mill	0.09	1.63	tons/acre/year
Line and White (2007)	North Carolina	0.16	2.80	tons/acre/year
Nelson and Booth (2002)	Washington	0.11	0.43	tons/acre/year
Owens et al. (2000)	Wisconsin – summer construction	0.18	3.38	tons/acre/year
	Wisconsin – winter construction	0.18	0.82	tons/acre/year
Reed (1980)	Pennsylvania –Site 2	0.17	8	tons/acre/year
	Pennsylvania – Site 2A	0.12	6	tons/acre/year
	Pennsylvania – Site 2B	0.21	7	tons/acre/year
	Pennsylvania – Site 3	0.20	9.5	tons/acre/year
Selbig et al. (2004)	Wisconsin – land disturbance phase	57.6	76.2	tons/storm event
	Wisconsin – house construction phase	9.7	10.4	tons/storm event

Table 4-3: Summary of Sediment Yield Data from Selected Studies

Study	Location	Sediment Yield: Unaffected Site	Sediment Yield: Affected Site	Units
Selbig & Bannerman (2008)	Wisconsin – major land disturbance phase	0.004	0.038-0.06	tons/acre/year
	Wisconsin – house construction phase	0.004	0.009	tons/acre/year
Vice et al. (1969)	Virginia	0.038	63-76	tons/acre/year
Ward and Appel (1988)	West Virginia	0.3	4.4	tons/acre/year
Wolman and Schick (1967)	Maryland - Baltimore	–	219	tons/acre/year
	Maryland - Towson	–	125	tons/acre/year
	Maryland - Cockeysville	–	112	tons/acre/year
	Maryland - Gwynns Falls	–	18	tons/acre/year
Yorke and Herb (1978)	Maryland – multiple sites		7.2-100.8	tons/acre/year

[1] Modeled estimate.

Table 4-4: Summary of Turbidity Findings from Selected Studies

Study Author	Location	Turbidity (NTU): Unaffected site	Turbidity (NTU): Affected site
Barrett, et al. (1995)	Texas	6	72
Chisholm & Downs (1978)	West Virginia	2.3	7.1
Embler & Fletcher (1983)	South Carolina	<25[1]	50- 80[1]
Line (2009)	North Carolina – Tilly Up	25	1,530
	North Carolina – Tilly Down	54	1,197
	North Carolina – Ellery Down	140	504
	North Carolina – King's Mill	41	593
Lubliner & Golding (2005)	Washington	2.2	2.2
	Washington	5.2	6.4
	Washington	5.5	5.2
	Washington	6.4	25.7
	Washington	9.8	45.0
	Washington	17.6	12.0
Shields & Sanders (1986)	Mississippi	18	40
Tsui & McCart (1981)	British Columbia	0.5-2.3	0.7-5,000
Wong (2005)[2]	Hawaii – Site 16226200	5.6	9.0
	Hawaii – Site 16265700	1.4	11
	Hawaii – Site 16266500	0.8	1.8
	Hawaii – Site 16267500	3.5	14
	Hawaii – Site 16269500	3.8	2.2
	Hawaii – Site 16270900	3.1	2.2
	Hawaii – Site 16273950	13	3.8
	Hawaii – Site 16275000	0.7	1.0

[1] Peak values.
[2] Wong (2005) measures concentrations as a long-term geometric mean.

4.2 Individual Study Summaries

This section provides brief summaries of studies of construction site discharges reviewed by EPA. Additional information is available in the original studies. Individual study summaries are presented in chronological order with the exception of state reports, which are summarized at the end of the section.

Whitney and Bailey (1959) documented large declines in fish abundance in a stream section that was channelized as part of a highway construction project. Small fish declined 85 percent in number and large fish declined 94 percent in number.

Guy and Ferguson (1962) described a lake in Virginia that accumulated approximately 235,000 tons of sediment between 1938 and 1957 due to construction and development taking place in 68 percent of the lake's watershed during this period.

Keller (1962) documented elevated suspended sediment discharge in a river in Maryland. Sediment concentrations and total flux were significantly higher at a monitoring station downstream of a rapidly developing area relative to those measured at a monitoring station in an upstream rural watershed.

Guy (1963) described a 58-acre residential construction site in Kensington, Maryland with an annual sediment yield of 39 tons/acre from 1957 to 1962. Suspended sediment concentrations from 1,490 to 65,000 mg/L were measured in the receiving stream. Early in the project, sediment from the construction site fully buried the receiving stream bed, which, under natural conditions, had a rock and gravel bottom. Much of the sediment was flushed from the channel by 1962, approximately 2 years after the peak of construction activity on the site.

Wark and Keller (1963), in a study of sediment sources in multiple Potomac River watershed subbasins, found the highest sediment yields in subbasins with significant levels of construction activity. Rates were 10 to 50 times higher than those in rural areas and varied with construction activity intensity. Annual sediment yields in watersheds of 4.1 to 72.8 square miles with large amounts of construction activity ranged from 1.7 to 3.6 tons per acre.

King and Ball (1964) documented impacts to a river in Michigan from highway construction. The highway project crossed eight major tributaries to the river at distances of 2 to 5 miles from the river. The authors observed elevated turbidity, suspended sediment, and sedimentation levels in the river. The sediments were primarily inorganic. Primary production declined 70 percent once construction activity began. Macroinvertebrate biomass declined. Smallmouth bass populations also declined, which the authors attributed to the sedimentation of the river's pools, important habitat for the bass. The authors also noted capture of sediments by patches of aquatic vegetation able to grow in the river because of nutrient discharges from a sewage treatment facility. The authors expressed concern that the combination of aquatic vegetation and sedimentation would create a mud flat ecosystem in place of the former coarse substrate of the river.

Wolman and Schick (1967) examined multiple studies conducted in the Mid-Atlantic region and found that sediment flux in streams receiving construction site discharges was two to several hundred times greater than that in streams draining rural or wooded areas. The authors found annual sediment yields of 18 to 219 tons per acre from construction sites 1.2 to 151 acres in size. The authors stated that, under non-construction conditions, annual sediment yields average 0.3-0.8 tons per acre, with lower levels (e.g., 0.02 tons per acre) in predominantly wooded watersheds. The authors noted that higher sediment yields were found in those watersheds with the greatest proportion of land disturbed by construction and the least "dilution" by lower sediment yields from other land uses. For example, the highest sediment yield noted in the study, 219 tons/acre/year, was associated with a drainage area of 1.2 acres that was completely disturbed by construction activity. The study noted sediment concentrations in construction-affected surface waters of 3,000 to greater than 150,000 mg/L, whereas the highest concentration found in waters draining predominantly natural or agricultural areas was 2,000 mg/L.

The study also described in detail the impacts of sedimentation from an individual construction site in Maryland. Virtually all stream sections contained sand or silt deposits, in contrast to natural conditions consisting of a predominantly cobble streambed and riffle and pool sequences. Portions of the stream bed had sand deposits up to 2 feet thick that traveled downstream in dunes. At all sites the authors examined, construction sediment impacts were visible for the entire length of the stream between the construction site and the stream's confluence with a reservoir (up to 2 miles). The authors noted that sediment from construction activity was nearly absent from a different stream in the same area seven years after the cessation of construction upstream. The authors did not describe the manner in which this sediment was redistributed downstream of the site of original impact. The authors noted the difficulty of predicting the duration and extent of sediment storage in different waters.

Vice et al. (1969) documented that highway construction constituted 1 to 10 percent of a 4.5 square mile Virginia watershed's area at any given time but contributed 85 percent of total sediment yield. Measurements were taken nearly 2 miles downstream from the construction area, indicating that the construction impacts were not solely localized to the active construction site. The authors also noted significant turbidity, which imparted a reddish color to the affected stream. Turbidity and suspended sediment concentrations increased as construction activity increased and abated as construction activity decreased. The stream carried a 90-day mean concentration of 20,000 to 25,000 mg/L of suspended sediment during precipitation events for several months.

Walling and Gregory (1970) documented a 2- to 10-fold and occasionally up to 100-fold increase in suspended sediment concentrations in waters draining areas undergoing construction.

Barton et al. (1972, as cited in Barton 1977 and Barrett et al. 1995) found that highway construction extirpated invertebrates in a Utah stream bottom. The stream bottom recolonized within 6 months of construction completion.

Peterson and Nyquist (1972) studied highway and bridge construction impacts on an Alaskan stream. The stream was still recovering from damage from past placer mining activity. The authors documented elevated turbidity and conductivity levels downstream of the site. No other water quality parameters changed (see *Table 4-1*). They also documented severe declines in the number and diversity of benthic macroinvertebrates downstream of the site. Macroinvertebrate abundance and density approached normal levels one year after construction.

Huckabee et al. (1975) documented impacts to a small stream from a highway construction project in Great Smoky Mountains National Park in Tennessee. Stream acidification took place shortly after completion of the project and resulted in a fish kill. Ten years later, the authors documented elevated sulfate, calcium, and metal (aluminum, magnesium, zinc, manganese, lanthanum, and samarium) levels and acidification in the stream due to iron sulfides in rock used for roadbed fill in 1963. Upstream of the site, stream pH was 6.5 to 7.0. Downstream of the site, stream pH was 4.5 to 5.9. In addition, a dense white-yellow precipitate coated stream substrate rocks and periphyton more than 1 mile downstream. The impacted stream was nearly devoid of fish (e.g., brook trout), salamander larvae, and macroinvertebrates for up to 4 miles downstream of the fill. Conditions improved gradually with increasing distance downstream. Fish killed by the stream water showed clear evidence of gill hyperplasia. The elevated acidity, sulfate, and heavy metal levels were documented as responsible for fish and salamander mortality. Kucken et al. (1994) documented that these conditions continued to persist 30 years after construction completion. Two species of stream-breeding salamander were nearly eliminated, and abundances of two other species were halved (Kucken et al. 1994).

Burton et al. (1976) documented impacts to a Florida stream and lake from highway construction. The authors documented increases in turbidity and suspended sediment concentrations, as well as in mean loads per storm for suspended solids, total dissolved phosphorus, dissolved silicon (silica), and orthophosphate. Loads of nitrate, nitrite, and ammonia also increased, but the authors postulated that these may have been associated with an upstream sewage discharges (even though these discharges affected both the upstream and downstream monitoring stations). The authors attributed the elevated levels of suspended solids, phosphorus, and dissolved silicon to soil erosion from the construction site. Total dissolved solids loads and total stormwater discharge did not change due to construction activity. The lake downstream experienced elevated turbidity and sedimentation levels.

Eckhardt (1976, as cited in Barrett et al 1995b) documented impacts from highway construction on a Pennsylvania stream. Suspended sediment discharge from the construction site comprised approximately 50 percent of the total suspended sediment load in the stream over a 3.5-year period. High sediment yields from the site continued after completion of construction because of delay in vegetation growth.

White (1976, as cited in Taylor and Roff 1986) documented increases in nitrogen and other inorganic ions in a stream downstream of a road construction site.

Barton (1977) studied highway construction site impacts to an Ontario stream and associated ponds up to approximately 1 mile downstream over the life of the construction project. The study documented elevated suspended sediment and sedimentation levels (see *Table 4-2*). Sedimentation levels increased 10-fold. The decline in stream sediment organic matter content reflected the erosion of low organic matter soil from the construction site. An increase in nitrate concentrations was observed during the final construction stages. Suspended sediment and sedimentation levels declined with increasing distance downstream due to sediment accumulation in several ponds along the stream channel, though elevated sedimentation levels were also seen downstream of the ponds after heavy spring flows. The number and standing crop of fish declined from 24 kg/ha to 10 kg/ha immediately downstream of the construction site during construction but returned to normal within 8 months of construction completion. The authors postulated that fish may have migrated from the area during construction to avoid high suspended sediment levels. No impacts on fish were documented further downstream, and no fish mortality was observed at any point. Benthic macroinvertebrate abundance and species number remained steady, but species composition changed. Areas completely denuded by construction activity were quickly recolonized by species known to be strong recolonizers. The authors postulated that some invertebrates may have sheltered in protected areas during periods of high sediment levels and others drifted in from unaffected areas. No changes were recorded in ammonia, dissolved oxygen, hardness, nitrite, pH, or phosphate levels.

Garton (1977, as cited in Barrett et al. 1995) documented an incident in which a highway construction site in West Virginia discharged large quantities of sediment to a karst cavern system. Springs discharging from the caverns were being used as the water supply for a fish hatchery. Sediment accumulation on the gills of trout in the hatchery killed 150,000 fish during one event. Other fish kills were the result of a diesel fuel spill on the site that subsequently washed into the caverns.

Reed (1977) documented impacts to several Virginia streams from highway construction. The authors documented declines in both macroinvertebrate and fish diversity and abundance due to elevated sediment discharges and consequent sedimentation from construction sites. Populations recovered somewhat after sediment discharges declined and excess sediments were flushed from the streams, though many had still not fully recovered several to 22 months following construction completion.

Chisholm and Downs (1978) studied highway construction impacts to Turtle Creek in West Virginia, also discussed in Downs and Appel (1986) (see below). Portions of the stream under study were in a relocated channel. The authors documented elevated turbidity, dissolved solids, sedimentation, and TSS levels (up to 11,100 mg/L). The authors recorded up to 8 to 10 inches of silt and clay on the stream bed. Sedimentation levels varied during the study period as excess sediments were flushed downstream by heavy winter and spring flows, followed by additional sedimentation during lower flow periods. The authors postulated that the elevated dissolved solids levels derived from use of calcium chloride as a wetting agent to reduce dust on the construction site during the early and middle phases of construction. Calcium chloride is highly soluble and leaches readily. Dissolved solids concentrations decreased toward the end of the construction project.

The authors also observed iron precipitates deriving from groundwater discharges. The precipitates elevated stream turbidity and imparted a reddish-orange color to it. The authors stated that such discharges are common in recently excavated areas in the watershed.

Heavy sedimentation levels and relocation of portions of the stream channel initially extirpated or heavily degraded macroinvertebrate populations over several miles of stream. Organism diversity and abundance recovered within a year after construction completion as stream substrate was flushed of excess sedimentation, then stabilized. The authors stated that organism drift from unimpacted tributaries upstream and good quality construction and revegetation of the new stream channels aided community recovery.

Extence (1978) documented elevated sedimentation levels and suspended solids and iron concentrations in a stream and river in Great Britain downstream of a highway construction site. Sedimentation proceeded to the point where fine sediment fully covered the former rocky substrate of the river. River channel width decreased as a consequence of sediment accumulation. High flow events periodically transported some of the sediments downstream. Sedimentation of the substrate reduced benthic invertebrate density and diversity. The authors attributed the elevated iron levels to erosion of iron-containing sediment from the construction site. The authors did not observe impacts to water column pH, BOD, orthophosphate, or nitrate levels.

Helm (1978, as cited in Barrett et al. 1995b) documented impacts from highway construction on a stream. Construction discharges contributed approximately 50 percent, or 8,000 tons, of the approximately 16,000 tons of suspended sediment discharged in the stream during the construction period. The stream already contained elevated sediment levels due to steep slopes, fine coal wastes, coal-washing operations, and other land uses in the upstream portion of the watershed.

Tan and Thirumurthi (1978) studied water quality changes in four lakes during highway construction activity taking place 30 to 400 feet from the lake shorelines. The lakes served as a drinking water supply for the city of Halifax, Nova Scotia. In one lake, elevated turbidity, total dissolved solids, and nitrate + nitrite-nitrogen levels were documented. Total dissolved solids were primarily composed of chlorides, sulfate, and sodium and calcium carbonates. Elevated conductivity was documented in two lakes. No water quality changes were documented in the other two lakes. *Table 4-1* lists the parameters for which no water quality changes were detected. The authors postulated that construction activity was the source of the water quality changes and that elevated nitrogen levels derived from the site's disturbed soils and vegetation. The authors stated that the highway projects had caused serious erosion, producing suspended sediment concentrations as high as 12,836 mg/L and very high turbidity concentrations in site runoff. The authors noted that sediment control devices treating the runoff to the primary affected lake performed poorly.

Yorke and Herb (1978) documented impacts to streams draining several Maryland watersheds over a multi-year period of development. The authors documented elevated suspended sediment yields from areas undergoing construction. The authors stated that an earlier study (Yorke and Davis 1972) had documented a 30 percent increase in stormwater runoff to streams due to construction taking place in 15 percent of the watershed.

Daniel et al. (1979) documented elevated yields of suspended sediment, total phosphorus, organic nitrogen, nitrate + nitrite, ammonia, and dissolved phosphorus from three residential construction sites in Wisconsin.

Hainly (1980, as cited in USFWS 2003 and Barrett et al. 1995b) found that highway construction site discharges elevated turbidity levels in a stream up to 6 miles downstream. The construction site contributed 9,100 tons of suspended sediment to the stream or more than 25 percent of the total suspended sediment load of 35,500 tons in the stream during the study period. The author observed little sedimentation immediately downstream of the construction site.

Reed (1980) monitored a highway construction site in Pennsylvania before, during, and after construction activity. The author documented elevated suspended sediment yields in four streams during construction. These yields fell approximately to pre-construction levels during the 2- to 3-year period following project completion. Suspended sediment concentrations also increased during construction and fell during the 2- to 3-year period following project completion.

Lenat et al. (1981) documented elevated suspended solids levels in the water column and increased fractions of sand and gravel in the substrates of two streams downstream of a construction project. These discharges impacted invertebrates in the stream. A suite of benthic macroinvertebrate species adapted to the new sandy substrate and associated periphyton downstream of the construction site was able to establish itself in high numbers during low flow periods. During high flow periods, however, this community was unable to persist. On average, organism density and habitat availability was lower downstream of the construction sites, but no species were entirely lost from the ecosystem. The authors noted that all stream sites in the study were impacted by discharges from upstream agricultural activities and gravel roads and so already contained a somewhat stress-tolerant benthic invertebrate community.

Tsui and McCart (1981) documented impacts to a stream in British Columbia due to construction of a gas pipeline crossing. Suspended sediment levels were elevated up to 10,660 mg/L and turbidity levels up to 5,000 NTU within 300 feet downstream of the construction site. Sedimentation levels increased. The authors documented a large reduction (up to 74 percent) in benthic macroinvertebrate abundance. Excess sediment was flushed downstream within 2 years of pipeline installation. The invertebrate community had not yet recovered after 2 years but was progressing in that direction.

Cline et al. (1982) examined stream sites within approximately 1,600 feet downstream of a highway construction project in a multi-year study. Construction activity elevated TSS levels 10- to 100-fold at downstream sites. The exception was one site where a reservoir release between the upstream and downstream sampling points influenced observations. TSS levels returned to background conditions within 2 weeks following cessation of construction. Construction also increased the percentage of fine sediments in the stream's substrate. This effect dissipated within 1 year of construction cessation, in large part due to high stream flow during the snowmelt period, which transported excess sediment downstream. Periphyton was generally less abundant and contained more detritus at downstream sites. The authors noted that fine sediment tended to collect in the periphyton. Downstream sites also had lower algal diversity. These effects were still apparent 1 year after construction completion. Macroinvertebrate density and biomass declined at some, but not all, downstream sites, and remained depressed for a year following construction completion. Taxa composition changed to favor more pollution-tolerant species. The authors postulated that construction activity timing had been such that high snowmelt flows and reservoir releases had transported sediments downstream and improved conditions at unaffected stream sites.

Cramer and Hopkins (1982, as cited in Barrett et al. 1995b) studied impacts to a Louisiana wetland from highway construction. The authors documented elevated turbidity and a change in water color deriving from construction discharges. Changes in salinity, dissolved oxygen, and pH were attributed to other factors. Turbidity decreased and water color began to return to normal levels once construction ceased.

Embler and Fletcher (1983, as cited in Barrett et al. 1995b) studied construction site impacts to a South Carolina stream. Turbidity levels increased from below 25 NTU to peak levels of 50 to 80 NTU. Suspended sediment levels increased from below 30 mg/L to peak levels of 60 to 130 mg/L.

Werner (1983, as cited in Donaldson 2004) documented a case in which improper highway construction practices created multiple instances of turbid water in karst springs in West Virginia. The turbid water clogged the gills of trout, killing a large number of them.

Helsel (1985) documented impacts to an Ohio stream and river from highway construction. Construction significantly elevated sediment yield and suspended sediment loads downstream.

Duck (1985) documented impacts to a stream and lake from gravel road construction in Scotland. A stream flowing into the lake carried a sediment load 20 times greater than normal. A turbid plume was observed to extend over 1,600 feet into the lake. The author documented deposition of at least 2,010 tons of sediment on more than 11 acres of lake bed during a 2-month period. The author estimated that the lake, under natural conditions, would have otherwise taken 20 to 25 years to accumulate this quantity of sediment. Suspended sediment concentrations in the affected streams returned to normal levels after completion of construction.

Downs and Appel (1986) documented construction site impacts to streams and rivers from highway construction in West Virginia. Suspended sediment loads in Trace Fork, a small stream in the study, increased from 830 to 2,385 tons downstream of the construction site for a 14-month period. Suspended sediment concentrations of up to 7,520 mg/L were measured in the stream. The largest daily mean concentration was 2,500 mg/L. Turtle Creek, another small stream, had elevated suspended sediment loads of 34,000 tons over 2 years. The authors were unable to gather sufficient data to determine impacts to suspended sediment in the Coal River, a river downstream of the smaller streams affected by construction activity. The authors postulated that the difficulty in measuring the impact to the larger river was due to masking of the construction discharge signal by dilution from the river’s large flow volume and by large sediment contributions from other sediment sources in the river’s watershed.

Nodvin et al. (1986) observed large sediment plumes extending across most of the length of a 2,200-foot wide lake whose inlet lay approximately 5.1 miles downstream of a bridge construction project. The sediment derived from preparation of an upstream area on a tributary stream for bridge footer emplacement. The authors documented elevated levels of alkalinity, conductivity, pH, and calcium in the stream. Maximum changes observed were 6.83 to 11.22 for pH, 9 to 341 micro-siemens per centimeter for conductivity, 70 to 2001 micro-equivalents per liter for alkalinity, and 54 to 2227 micro-equivalents per liter for calcium. Smaller increases in magnesium, sodium, and potassium levels were also recorded.

The authors postulated that leaching of calcium oxide hydration products from the cement in the concrete poured for the bridge footers was the pollutants' source. Conductivity and alkalinity were found to be elevated as far as the lake outlet approximately 5.6 miles downstream. Elevated pollutant levels ceased and dispersed within the 24 hours following the concrete pouring. The authors postulated that the short-term duration of the effects explains why the phenomena they documented have not been more widely reported in the literature.

Shields and Saunders (1986) documented stream impacts from waterway (canal) construction, a portion of which required stream rechannelization. The authors recorded elevations in specific conductance, turbidity, color, COD, total alkalinity, hardness, ammonia, phosphorus, sulfate, iron, lead and manganese during construction. Mean values were 50 to 100 percent higher than levels measured before the start of construction. Concentrations of dissolved calcium, dissolved magnesium, chloride, total nitrogen, total nitrite, and total Kjeldahl nitrogen also increased. Stream temperature and average daily loadings of total metals (including iron, manganese, lead, and zinc), nutrients, and dissolved solids increased. Coliform bacteria densities decreased. The authors ascribed these changes to increased sediment input from the construction site and reduced shading of the surface water due to removal of riparian vegetation during construction. Total organic nitrogen, phosphorus, zinc, iron, and manganese were all significantly correlated with turbidity discharges. Average daily suspended solids loadings remained approximately the same, though the authors ascribed this result to water quality sampling issues. The authors attributed the water quality changes to erosion from the construction site, including excavations that exposed formerly deeply buried materials to precipitation and erosion.

Taylor and Roff (1986) continued study of a small stream in southern Ontario impacted by both highway and sewer trunk line construction, extending the work documented in Barton (1977) (summarized above). Monitoring took place for six years following completion of highway building activities and included the land stabilization and revegetation phase of construction activity. During this period, part of the downstream study area was additionally impacted by sewage line construction.

The authors documented elevated suspended sediment and sedimentation levels downstream of the construction site that declined over several years as vegetation stabilized the former construction site and excess sediment migrated downstream. Site revegetation, in particular, reduced sediment discharges. The proportion of organic material in the suspended and bedded sediments increased over time as predominantly inorganic sediment from the construction site was flushed downstream. Stream sediment conditions closest to the construction site returned to reference levels approximately 5 years after construction site stabilization. Sediment conditions further downstream, however, did not fully recover during the study period due to additional disturbance from sewage line construction and probable input of sediments flushed downstream from sites closer to the original construction site. The stream bed remained covered by up to 6 inches of inorganic sediment. The authors documented extensive growths of previously absent aquatic plants downstream of the construction site, which they attributed to elevated sedimentation levels.

Elevated nitrate levels were also detected downstream of the construction site. Levels were higher in areas closer to the construction site and slightly lower at sites further downstream. The authors noted that phosphorus and nitrogen fertilizers were applied with straw mulch as part of the highway site stabilization process and that heavy rains washed much of the initial mulch application and, most likely, much of the fertilizer from the site (site culverts were clogged with straw). Elevated phosphorus levels were not detected.

The authors documented that the changes in stream conditions favored some species of invertebrates and fish and decreased or failed to favor others. For the first 2.5 years of the study, macroinvertebrate species tolerant of sedimentation and low-level organic enrichment increased in abundance and biomass. The authors postulated that the macroinvertebrates were responding to nutrient-enriched conditions in the stream deriving from construction site nutrient discharges. Species less tolerant of sedimentation increased in abundance more slowly or declined. Their loss led to an overall decline in community diversity. The magnitude of modification to the macroinvertebrate community declined with distance downstream from the construction site. In the final 3 years of the sturdy, diversity increased as excess sediments were flushed from the stream and formerly declining species populations strengthened.

Fish abundance increased downstream of the construction site during the study period. The authors postulated that this effect was due to the elevated abundance of macroinvertebrates and the growth of aquatic plants. The authors noted that fish populations declined downstream of the sewage line construction site, probably due to migration of fish from the area to avoid high suspended sediment levels. Population levels rebounded once construction was complete. Community composition changed to favor mid-water feeder species over bottom feeder species.

The authors postulated that several of the changes to the stream community were “relatively permanent” and that return to original conditions would take many years (assuming no further disturbance from new construction sites).

Ward and Appel (1988) continued the study initially described in Downs and Appel (1986) (see above). Over a period of nearly 5 years, the Trace Fork watershed saw completion and stabilization of construction activity for one highway segment, a 2-year period without construction, followed by a second construction phase. Sediment loads in Trace Fork were elevated during the first construction phase, declined during the inactive phase, and increased again during the second phase of construction.

Stout and Coburn (1989) studied macroinvertebrates that consume leaf detritus in a Tennessee stream 2 years after completion of highway construction upstream. Observation sites were located within 0.5 miles of the former construction site. The authors found stream substrate downstream of the construction site to have returned to conditions similar to those of upstream sites. The total abundance and number of macroinvertebrate taxa was lower downstream of the former construction site than expected. The authors attributed this observation not to construction sedimentation impacts but to the removal of riparian vegetation during construction, which reduced detrital inputs to the stream and elevated stream temperature.

Yew and Makowski (1989, as cited in Barrett et al. 1995b) documented impacts to several streams near the Tennessee-North Carolina border from highway construction. Exposure of pyritic shale during construction resulted in discharge of sulfuric acid to the streams. Conditions toxic to fish existed in the streams due to low pH levels of 4.0 to 4.4, low alkalinity, and elevated toxic metal concentrations.

Young and Mackie (1991) studied oil pipeline construction impacts to a stream in the Northwest Territories, Canada. Part of the construction process involved burying a portion of the pipeline beneath the stream. Downstream sampling stations were located within 1,500 feet of the construction zone. The authors documented elevated TSS concentrations downstream of the construction site that peaked at 3,000 mg/L. These sediments contained significantly less organic matter than the usual stream sediments. Macroinvertebrate drift increased as suspended sediment concentrations increased. The authors attributed this effect to the increased levels of sediment on the stream bed and in the water column. Total biomass and species richness recovered by the warm season following completion of construction. Sedimentation also increased downstream of the construction site, but excess sediments were flushed downstream by the end of the warm season following construction completion. Water chemistry parameters did not change during the study.

Barrett et al. (1995) studied highway construction impacts to a stream in Texas. The authors calculated percentage changes in concentrations of a number of parameters (see *Table 4-1*) but did not assess the statistical significance of the observed differences. Large increases in concentrations of TSS, turbidity, and iron were observed downstream of the site. Moderate increases were observed in concentrations of zinc, volatile suspended solids, and nitrate. Smaller increases were observed in copper, COD, total coliform, and fecal strep. Small declines were observed in concentrations of BOD5, fecal coliform, and total organic carbon. No change was observed in cadmium, chromium, or lead concentrations. The authors found iron discharges to be highly correlated with TSS discharges from the construction site but were not able to identify a source for the elevated zinc discharges. The authors observed elevated sedimentation levels in the stream downstream of the construction site. This excess sediment was flushed further downstream by storm flows within approximately 1.5 years following the cessation of construction.

Stephens et al. (1996) stated that road construction increased loadings of sediment, nutrients, and trace elements to a receiving stream and downstream reservoir used as a drinking water supply.

Welsh and Ollivier (1998) documented deposition of sediment from a California highway construction site into a stream. A single storm event deposited sediment layers 0.1 to 2 inches thick on the substrate of several streams within 0.5 to 1 mile of the construction site. Mean fine sediment depth and percentage of substrate embeddedness was significantly greater in impacted stream pools. The sediment filled interstitial spaces in the stream's substrate and significantly reduced the density of two salamander species and one frog species.

Reid and Anderson (1999) reviewed 27 studies of water quality impacts from pipeline stream and river crossing construction at various sites in North America. Pipeline crossing construction frequently requires construction activity directly in a surface water. Most earthmoving activity takes place over the course of a few days to a few weeks, though site stabilization can require additional time. Impacts varied among sites. Multiple studies documented increases in downstream levels of turbidity, suspended sediment, sedimentation, and stream embeddedness downstream during and after construction. Changes in channel morphology and substrate composition were also observed. Turbidity levels up to 4,700 NTU and suspended sediment levels up to 11,600 mg/L have been documented. Multiple studies also documented reductions in benthic invertebrate abundance and diversity and reductions in fish abundance. Invertebrate community compositions generally shifted to include a larger proportion of pollution-tolerant species. Decreases in invertebrate abundance and changes in community composition were attributed to increased drift downstream and habitat degradation. The authors also found some studies documenting little or no impact to fish communities.

Of those studies that looked at post-construction recovery, multiple studies documented recovery of downstream areas to pre-construction conditions within 2 to 4 years following installation of the crossing. Turbidity and suspended sediment levels typically decreased once major earthmoving activity was complete. Excess sediments were usually flushed downstream within 6 weeks to 2 years. Recovery of in-stream cover (pools, rocks, woody debris, and submersed vegetation) took 4 years in one study. Invertebrate communities moved toward recovery as excess sediments were removed and recolonization from undisturbed areas upstream took place. Fish populations also moved toward recovery with improvements in sediment conditions and invertebrate communities.

Owens et al. (2000) studied sediment yields in stormwater prior to its on-site treatment from one small commercial and one small residential construction site in Wisconsin. The authors found that construction activity significantly increased sediment yields from the sites relative to sediment yields from undisturbed rural areas (see *Table 4-3*). Average event mean concentrations of suspended sediment also increased significantly. Increases in sediment yield and event mean suspended sediment concentration were less at the site that was active during winter, when rainfall events were less intense, than at the site that was active during the summer, when rainfall events were more intense.

Fossati et al. (2001) studied a river in Bolivia impacted by highway construction. The small river downstream of the construction site was transformed by excess sediments into a braided stream system. The authors documented a 500-fold increase in suspended sediment concentrations, which ranged from 2,574 mg/L near the construction site to 5 to 267 mg/L further downstream. Turbidity impacts were observable 56 miles downstream. Smaller increases in phosphate and bicarbonate concentrations were observed. Sedimentation levels increased significantly. Aquatic invertebrate density declined 200-fold, and the total number of taxa declined 6-fold. Authors attributed the invertebrate decline to various impacts associated with increased suspended sediment concentrations. No changes in pH, temperature, or concentrations of chlorides, sulfates, nitrates, calcium, magnesium, sodium, or potassium were observed.

Hunt and Grow (2001) documented impacts to a high quality stream downstream of a construction site that failed to comply with discharge requirements. Sedimentation created a silt and clay layer 12 to 16 inches thick on the stream bed. The stream's substrate, gravel and cobble under natural conditions, was highly embedded 5,000 feet downstream. Number of fish species decreased from 26 at the upstream site to 19 at the downstream site. Fish abundance decreased from 525 to 230 individuals. Pollutant-tolerant species increased 237 percent and pollutant-intolerant species decreased 33 percent. Fish biomass declined 62 percent. Measures of overall habitat quality significantly declined. The authors attributed the decline in aquatic community health to the severe embeddedness of the stream's substrate.

Kayhanian et al. (2001) studied 15 California highway construction sites with widely varying characteristics over a 2-year period. In stormwater discharges from the sites, the authors documented varying levels of particulate cadmium, nitrate, nitrite, ammonia, total Kjeldahl nitrogen, suspended and dissolved solids, turbidity, chemical oxygen demand, oil and grease, total and fecal coliform, chlorpyrifos, diazinon, and dissolved and particulate chromium, copper, lead, nickel, silver, zinc, and phosphorus. The authors attributed TSS and turbidity levels runoff to disturbed soils on the sites. They stated that the sources of the other pollutants were unknown. Because levels of these pollutants varied among sites, the authors speculated that site-specific soil and vegetation conditions influenced their discharge. The authors also noted correlations between TSS concentrations and particulate copper, chromium, and zinc concentrations, indicating that these pollutants are traveling with sediment during erosion.

Buckner et al. (2002) stated that sediment is the top source of impairment in the Tennessee, Cumberland, and Mobile River basins, a region containing high levels of aquatic organism diversity. The study noted that a large number of aquatic species in this region are threatened, endangered, or being considered for listing. Construction activity is one source of sediment in the region. The authors described an incident where improper control of runoff from a road construction site in Tennessee resulted in stream vegetation loss and stream bank collapse. Four feet of sediment accumulated on the stream bottom, extirpating the existing biological community. The water supply for a downstream town was contaminated. The authors also noted a report of significant levels of sedimentation in the Cahaba River due to sewer trunk line and residential home construction.

Nelson and Booth (2002) noted that construction sites had one of the highest sediment yields per unit of land area of all land uses in western Washington, approximately four times greater than natural sediment yield for the region. Construction comprised 0.3 percent of the area of the watershed under study and contributed 0.7 percent of total watershed sediment yield.

Carline et al. (2003) documented statistically significant increases in TSS concentrations and suspended sediment loads at sampling sites within 1,200 feet downstream of two different road construction sites. The authors monitored the site during the entire construction process. Total TSS export at one site increased at least 13.5 percent relative to upstream levels after installation of concrete-lined drainage channels on site. The authors noted that interpretation of data from this site was complicated by probable elevated sediment export from an additional construction site upstream of the observed construction site. Total TSS export at the second study site was 13.6 percent higher. The authors did not find impacts to stream substrate composition, benthic macroinvertebrate communities, or trout redd occurrence downstream of either site.

Selbig et al. (2004) studied discharges from a residential construction site over the life of the project. The authors found that discharge volumes and total and suspended sediment loads were significantly greater in the stream downstream of a residential construction site compared to upstream of the site. The authors noted that careful implementation of a wide range of sediment and erosion control practices significantly reduced discharge volumes and sediment loads during the life of the project. The authors detected no significant impacts on stream geomorphology, macroinvertebrate communities, and fish communities downstream of the site. However, the stream drained a heavily agricultural area and was ranked very good to good for macroinvertebrate community quality, poor for fish community quality, and fair for overall habitat quality prior to the start of construction activity. Changes in temperature and fine sediment levels were attributed to processes unrelated to construction activity.

Lubliner and Golding (2005), in a study designed to examine a wide range of construction sites, visited 183 sites in Washington. They found 44 to be discharging runoff at the time of their visit. Other sites were not discharging due to the episodic nature of precipitation events, lower than normal rainfall, on-site infiltration, and use of stormwater management practices. Of the 44 discharging sites, 6 discharged directly to surface waters and 38 infiltrated discharges or directed them to a municipal stormwater sewer. Two of the six sites' discharges elevated turbidity levels 100 feet downstream. The authors stated that the low incidence of receiving water turbidity impacts might have reflected the lower level of precipitation during the study and efforts at construction sites to avoid discharging to surface waters.

Wong (2005), in a multi-year study, documented elevated total sediment yield, TSS concentrations, and turbidity levels at multiple stream sites within approximately 0.5 miles of a highway construction project that were most likely attributable to construction activity. Elevations were seen in both long-term geometric means as well as 90th and 98th percentile values. Some sites were also disturbed by golf course construction. TSS and turbidity levels declined after completion of construction except for TSS levels at two sites and turbidity levels at one site. Elevated nitrite + nitrate, total phosphorus, copper, and specific conductance levels were also thought to derive from construction activity, but were detected at substantially fewer stream sites. No effects from construction activity were seen on a reservoir approximately 0.5 miles downstream of the project.

Cleveland and Fashokun (2006) documented a two- to six-fold increase in TSS concentrations due to highway construction activity.

Hedrick et al. (2006) found no statistically significant difference in water quality between sites 0.2 km downstream of construction activity and unaffected sites. The authors speculated that the lack of elevated suspended sediment levels was due to frozen soil conditions during the peak construction period, decreased rainfall intensities, and/or effective sediment erosion and control practices. The authors also noted that it was likely that the construction site elevated sediment discharges to the stream, but that they were difficult to detect statistically because the construction site constituted a small portion of the stream's total watershed, site discharges were diluted approximately 3,300-fold in the stream, and the stream's suspended sediment concentration was naturally highly variable.

Hedrick et al. (2007) examined impacts from a West Virginia highway construction project during a 3-year period at two stream sites 300 to 800 feet downstream. The construction site significantly increased total sediment export downstream, particularly prior to the installation of silt fences. During this time, benthic macroinvertebrate community health declined in one stream. Community health recovered once silt fences reduced sediment export from construction activity. Over the total duration of the study, construction activity did not have a significant impact on the benthic macroinvertebrate community's health. The authors postulated that stream areas downstream of the construction site were successfully recolonized by organisms from unaffected stream areas. The authors also found little difference in the percentage of fine sediment present at upstream versus downstream sampling sites.

Clausen (2007) documented changes in stormwater runoff quality in storm sewer and ditch discharges from two watersheds under construction, one using traditional sediment and erosion control practices and the other using more rigorous ("enhanced") control practices. Though runoff volume typically increases during construction activity, it declined 97 percent from the enhanced BMP watershed. The authors attributed this change to sediment and erosion control practices that retained water on-site and the placement on-site of soil with higher infiltration values than the site's original soil. Runoff volume doubled in the traditional watershed. In the enhanced BMP watershed, both TSS and phosphorus total export and concentrations increased. Total export of nitrogen stayed the same, though nitrogen discharge concentrations increased. The author attributed increased nitrogen concentrations to fertilizer use. In the traditional watershed, total export of both sediment and nutrients increased. TSS concentrations did not change, which the author attributed to adequate sediment control practices. Nitrogen and phosphorus concentrations stayed the same or declined. Zinc export from the enhanced BMP watershed declined during construction, and copper and lead export stayed the same. Metals export from the traditional watershed increased. Fecal coliform and BOD levels did not change.

Line and White (2007) compared stormwater runoff immediately downstream of an area undergoing residential development to a nearby undeveloped area. Construction activity elevated the percentage of rainfall leaving the area as stream runoff to 55 percent versus 21 percent at the unaffected site. Runoff volume was 68 percent greater. The site elevated TSS, total phosphorus, nitrate, TKN, ammonia, and total nitrogen yields (see *Table 4-1*). Stream baseflow declined to 0 percent of flow versus 25 percent at the unaffected site and peak discharge rates increased more than four times.

Selbig and Bannerman (2008) found that runoff volume from an area undergoing residential construction in Wisconsin increased relative to pre-construction conditions. The authors documented that the Low Impact Development (LID) techniques used to develop the area, however, had significantly reduced runoff volume relative to conventional development techniques. TSS, total solids, and total phosphorus yields also increased during construction.

Chen et al. (2009) found statistically significant increases in TSS and turbidity levels at stream sites approximately 100 to 3,400 feet downstream of construction activity. The authors noted that the magnitude of the increase in TSS and turbidity levels was less than that recorded in many other studies and thought this was possibly due to a combination of sediment and erosion controls at the construction site and the predominance of woodland and pasture in the contributing watersheds. Total iron, sulfate, chloride, and potentially other water quality parameters were elevated during construction activity, and chloride, sulfate, nitrate, and acidity were elevated after project completion. The authors thought the nitrate may derive from the use of mulches, fertilizer, and hydroseeding for erosion and sediment control at the construction site. The authors documented a decline in overall stream ecosystem health as indicated by the percentage of chironomidae, the Hilsenhoff Biotic Index, and Ephemeroptera, Plectoptera, and Trichoptera (EPT) presence. Stream health remained good to excellent overall during and after construction, however, which authors partially attributed to the overall wooded and rural nature of the affected watershed and to the possibility that the stream may not have exceeded certain thresholds for fine sediment accumulation critical to macroinvertebrate health.

Lee et al. (2009) documented substantially higher sediment yields per unit area from watersheds with higher levels of road and building construction activity compared to watersheds with lower levels of activity. Sediment yields were 1.6 tons/acre/year for the watershed with the greatest level of construction activity and fell as low as 0.2 tons/acre/year for already urbanized or less-developed watersheds. The author also observed elevated levels of fine sediment deposition in areas with high levels of construction activity. The watershed under study was listed as biologically impaired due, in part, to excess suspended sediments.

Line (2009) described impacts to streams and a lake from highway construction in North Carolina. The lakes were located in a residential neighborhood that valued them for their aesthetic and recreational values. Construction discharges elevated suspended sediment and turbidity levels in tributary streams and the lake. One watershed (“Ellery-down”) was also impacted by discharges from widening of a municipal road and residential construction. Larger increases in suspended sediment yield were found in watersheds that contained a larger proportion of construction activity and/or contained construction activities particularly prone to erosion.

Montgomery County Department of Environmental Protection (2009) is the most recent annual report for a multi-year study of impacts to surface waters from construction and land use change on previously undeveloped lands in the county. The report described changes seen in three large watersheds undergoing development. Enhanced sediment and erosion controls were in use and generally performed as expected. Some monitoring was conducted immediately downstream of construction sites, while other monitoring was conducted further downstream in order to examine impacts deriving from larger areas containing multiple construction projects whose locations changed over time. The report noted that impacts were in part deriving from disturbed lands where construction had started but was subsequently halted due to an economic downturn. Disturbed lands were left in an unstabilized condition. Stream chemistry was monitored downstream of one construction project. TSS concentrations increased during construction, declined during site stabilization, and returned to pre-construction levels during the post-construction period. TKN levels may have increased when sediment and erosion control structures from the construction phase were converted to permanent stormwater control structures. Substrate embeddedness impacts were documented downstream of two of six construction projects during 2007. Embeddedness levels improved to pre-construction levels once construction was completed. Another stream underwent channel aggradation and straightening during three years of active construction in the watershed. Total water discharge to the stream increased during this time, and flow was “flashier.” As construction activity declined in the watershed during two to three years, the channel started to erode accumulated sediments and to widen. Flow became less flashy. Calculation of Index of Biological Integrity scores for benthic macroinvertebrates and fish for multiple stations throughout the three large study watersheds indicated degradation of aquatic ecosystem health from construction activity. Scores for most areas dropped from "good" or "excellent" to "fair" or "poor." Areas with more intense construction activity scored more poorly. The authors noted a shift in macroinvertebrate community structure to include more generalist and pollution-tolerant organisms. One area of one watershed showed indications of moving toward recovery of pre-construction conditions.

4.3 State Reports of Construction Discharge Impacts

A number of states have identified construction as a source of damage or impairment to surface waters in written reports and in required reporting of impaired waters under Sections 303(d) and 305(b) of the Clean Water Act.

Ohio's state water quality reports from the early 1990s through 2006 include 29 reports identifying water quality degradation due to construction activity (Ohio EPA 1994, 1995, 1996a, 1996b, 1996c, 1997a, 1997b, 1997c, 1997e, 1997f, 1998a, 1998b, 1998c, 1999a, 1999b, 1999c, 1999d, 2000, 2001b, 2003a, 2003b, 2004, 2005, 2006a, 2006b, 2006c, 2006d, 2007b, 2007c). Water quality reports from Ohio EPA evaluate biological indicators of water quality and, where possible, identify the waterbody's sources of impairment. Multiple reports identify fish and macroinvertebrate communities impaired by elevated suspended sediment and/or sedimentation levels from construction activity upstream. Several types of construction are identified as sediment and/or turbidity sources in the studies, including road (Ohio EPA 1997e, 1998c, 1999b, 2001a), bridge (Ohio EPA 2001a, 2001b), highway (Ohio EPA 1994, 1998b, 1998d, 1999d, 2005, 2005, 2007a, 2007b), residential (Ohio EPA 1997d, 2006d), commercial (Ohio EPA 2005), golf course (Ohio EPA 1997f, 1998e, 1999d, 2004, 2006b, 2007c), and airport (Ohio EPA 1996a, 1998c, 2001b).

Ohio EPA (1997f) documented impacts to a stream from a combination golf course and residential housing construction site. Elevated turbidity and sedimentation were observed up to 10 miles downstream. Within a short distance downstream of the site, dissolved oxygen concentrations were depressed, and TSS, fecal coliform, ammonia, and nitrate+nitrite concentrations were elevated. The stream bed was covered with 4 to 8 inches of silt. The authors stated that the construction site was the primary source of the pollutants but upstream agricultural sources also contributed. Discharges from the site degraded stream habitat. Approximately 100 dead fingernail clam (*Musculium transversum*) individuals were found in a downstream pool, and the macroinvertebrate community consisted of pollution-tolerant species. Healthy populations of fingernail clams and pollution-sensitive macroinvertebrates were found upstream and several miles downstream of the site. The fish community was also somewhat impacted.

Ohio EPA documented road construction sites as possible contributors to elevated polycyclic aromatic hydrocarbon (PAH) levels in at least three rivers: Little Cuyahoga River, Little Miami River, and Scioto River (Ohio EPA 1998d, 1998e, 1999b). PAHs are found in asphalt and tar used in road construction and repair, and it is possible for them to contaminate sediment and construction site discharges during rain events (Ohio EPA 1998d). Ohio EPA also attributed increased chromium concentrations in receiving waters to pressure-treated wood products used at a nearby residential construction site (Ohio EPA 1997d).

The state of California has expressed concern about discharges from construction sites and has specifically discussed the impacts of these discharges on threatened and endangered California coho salmon and steelhead trout and their habitat (McEwan and Jackson 1996; California Department of Fish and Game 2004). The state noted, for example, that discharges from the construction of Interstate 5 in California depleted spawning gravels for coho salmon (California Department of Fish and Game 2004).

The states of Michigan and Missouri have also issued reports that identify construction as a source of water quality degradation (Missouri DNR 2004, 2005; Michigan DEQ 2006).

Table 4-5 summarizes data from the ATTAINS database, a national database of water quality information submitted to EPA by all states, territories, and the District of Columbia (see *Section 2.6,* for more information). This table presents information on surface waters identified as impaired due to construction activity as of September 17, 2009. Though substantial, the numbers in *Table 4-5* probably underestimate the level of impairment due to construction activity because states are not required to identify surface water impairment sources. As of September 17, 2009, 29 states include “Construction” on their lists of possible impairment sources. Because significant effort can be necessary to identify a surface water’s source of impairment, it is likely that a number of construction sites contributing to surface water impairment have not been identified as impairment sources. In addition, as discussed in *Section 2.6*, a large percentage of surface waters have not yet been assessed for impairment.

Table 4-5: Construction Impairment in 305(b)-Assessed Waters

Stream/ River (miles)	Lake/ Pond/ Reservoir (acres)	Bay/ Estuary (sq. miles)	Coastal Shoreline (miles)	Ocean/ Near Coastal (sq. miles)	Wetland (acres)	Great Lake Shoreline (miles)	Great Lake Open Water (sq. miles)
12,273	300,855	16	3	3	13,971	–	–

Source: EPA ATTAINS database (EPA 2009a) as of 9/17/09.

5 Overview of Benefits from Regulation

This chapter provides an overview of the potential benefits to society related to the reduced sediment turbidity and reduced nutrient discharges from construction sites that will result from the final regulation. Sediments and other pollutants from construction sites (e.g., nutrients) may have a wide range of effects on water resources located in the vicinity of construction sites. These environmental changes affect economic productivity (e.g., drinking water supply and storage and navigation) as well as environmental services valued by humans (e.g., recreation, public and private property ownership, existence services such as aquatic life, wildlife, and habitat designated uses). Related market benefits (e.g., avoided costs of producing various market goods and services) and nonmarket benefits are additive (Freeman 2003). In all cases, benefits are conceptualized and estimated based on established welfare theoretic models (Freeman 2003; Just et al. 2004).

This chapter provides a conceptual framework for understanding the benefits likely to be achieved by the regulation, a qualitative discussion of those benefits, and a review of the benefit estimation methods used in EPA's analysis of this regulation. The following chapters quantify and estimate the economic value of these benefit categories in greater detail.

5.1 Conceptual Framework for Valuation of Environmental Services

The economic benefits of the final regulation reflect the money-metric values associated with the resulting improvements in water and other resources. The conceptual approach for estimating the monetary value of benefits of a policy involves an evaluation of changes in social welfare realized by consumers and producers, and quantification of these changes in money-metric terms. Such measures are based on standardized and widely accepted concepts within applied welfare economics (Just et al. 2004). They reflect the degree of well-being derived by economic agents (e.g., people and firms) given different levels of goods and services, including those associated with environmental quality. For market goods, analysts typically use money-denominated measures of consumer and producer surplus, which provide an approximation of exact welfare effects (Freeman 2003). For nonmarket goods, such as aquatic habitat, values must be assessed using nonmarket valuation methods (Freeman 2003). In such cases, valuation estimates are typically restricted to effects on individual households (or consumers), and either represent consumer surplus or analogous exact Hicksian welfare measures (e.g., compensating variation). The choice of welfare measure (i.e., value) is often determined by the valuation context.

The economic value of the changes in environmental services provided by surface waters from the regulation is a product of three sets of functional relationships (Freeman 2003):

- The effect of the regulation on environmental quality *(Q)*
- Human uses of the affected resources and their dependence on environmental quality *(X)*
- The economic value of the uses of the environment *(V).*

The first functional relationship relates the final regulation to changes in environmental quality. In this case, water quality is a function of sediment and other pollutant loadings *(S)* from construction sites, in-stream pollutant concentrations (vector *P),* and other relevant attributes (vector *A)* such as stream flow and velocity. Because the final regulation affects private activities that influence sediment and other pollutant discharges from construction sites, pollutant loadings are a function of private responses to the final regulation *(R).* This relationship may be represented as:

$$Q = f(S(R), P, A) \quad \text{(Eq. 5-1)}$$

For this analysis, water quality changes are estimated in terms of sediment and nutrient concentrations in receiving waters using SPARROW, an empirical relationship between sediment and nutrient concentrations in surface waters, and DCP. (see *Chapter 4* and *Appendix C* through *Appendix F* for details).

The second set of functional relationships describes the human values of the affected water resources and their dependence on water quality. In the following analyses, pollutant discharges are treated as an "input" to the water quality, and, as a result, as an "input" to ecological services provided by the affected water bodies. Like any other input, the value of changes in pollutant discharges from construction sites is determined by its impact on water quality. Water quality, in turn, influences human uses of the affected resources, leading to changes in use values. It may also lead to changes in ecosystem services that generate nonuse values.

For example, water quality is often characterized based on its suitability for recreational activities (e.g., whether the water is boatable, fishable, or swimmable) or its ability to support specified uses (e.g., primary/secondary contact recreation, public drinking water supply, or commercial fishing). To link water quality changes from reduced construction site discharges to effects on human uses and support for aquatic and terrestrial species habitat, EPA used a Water Quality Index (WQI) for this analysis (McClelland 1974; Dunnette 1979; Harrison et al. 2000; CCME 2001; Cude 2001; Carruthers and Wazniak 2003; Gupta et al. 2003; USEPA 2004b). *Section 10.1* provides detail on the WQI and its application to the benefits analysis for the regulation. The WQI presents water quality in a way linked to suitability for varying human uses, but does not in itself identify associated changes in human behavior. Behavioral changes and associated welfare effects are implied in the proposed benefit transfer approach for measuring economic values. Additional details of this approach are provided below and in the following chapters.

In addition to water quality *(Q),* the level of human uses or other ecological services (vector *X)* of the environmental resource depends on a variety of other factors (e.g., type and physical characteristics of the waterbody, proximity to populated areas, presence of recreational amenities, scenery and etc). If vector *I* represents these other inputs into the production of environmental services or uses based on the affected water resource, the second functional relationship can be expressed as:

$$X = f(Q, I) \quad \text{(Eq. 5-2)}$$

The vector of environmental services or human uses *(X)* affected by changes in water quality and sediment discharges could include services received through purchases of market goods produced with surface water resources (e.g., commercial fish), goods, services, and activities produced within the household, such as water-based recreation, and existence values associated with water quality improvements.

The third set of relationships defines the economic value *(V)* of the ecological services or human uses of the environment. This relationship "embodies the value judgment society has adopted for economic purposes" (Freeman 2003). For simplicity, and following standard approaches in benefit cost analysis

(Boardman et al. 2001), this analysis assumes that the value function is a simple aggregation of individuals' values.[2]

The relationship between individual or household utility *(U)* and water quality *(Q)* can be more formally described though the following utility function:

$$U = U(X(Q), Y, B) \tag{Eq. 5-3}$$

Individuals are assumed to derive utility from *X,* as well as from other goods and services *(Y),* which are not related to water resources. For example, a subset of household goods and services, such as recreation, are "produced" with water quality *(Q)* and other inputs such as travel. To account for variation in this utility structure across individuals, *B* denotes a vector of individual or household characteristics, such as demographic characteristics, that influence the form or parameters of the utility function.

Maximization of Equation 4-3 with respect to an income *(M)* constraint produces the following indirect utility function:

$$V = V(Q, P, M; B) \tag{Eq. 5-4}$$

where *P* represents the vector of prices associated with goods and services *(X and Y).*

The total dollar value associated with a change in water quality from Q_0 to Q_1 (holding other variables constant) can therefore be expressed using the compensating variation*(CV)* measure from the following equation:

$$V(Q_0, P_0, Y_0; B) = V(Q_1, P_0, Y_0 - CV; B) \tag{Eq. 5-5}$$

Other things being equal, an individual is better off with a higher level of water quality (Q_1) than a lower level (Q_0). CV in this equation represents the individual's maximum willingness to pay for the specified water quality improvement, leaving the individual just as well off after payment and higher water quality as with lower water quality.

From the above, CV (or willingness to pay, WTP) may be derived as a function of pre- and post-change environmental quality, prices, income, and individual attributes:

$$CV = CV(Q_0, Q_1, P_0, Y_0; B) \tag{Eq. 5-6}$$

This function also provides the basic conceptual foundation for constructing a benefit transfer function, which can be used to predict WTP values for defined changes in water quality (van Houtven et al. 2007).

5.1.1 Applicable Benefit Categories

There are likely to be several categories of benefits of the regulation; some benefits are linked to direct use of market goods and services, and others pertain to nonmarket goods and services. The following sections outline the most important categories of the expected benefits, and approaches used to estimate each category. In cases where estimates were not possible due to data limitations or other constraints, this is also noted below.

2 This function could also incorporate weights or environmental justice concepts.

5.1.2 Market Benefits

Some of the social benefits expected from the regulation will manifest themselves in economic markets through changes in price, cost, or quantity of market-valued activities. For benefit endpoints traded in markets, such as increased yields from commercial fisheries, benefits can be measured using standard market approaches based on data including market prices and quantities (Boardman et al. 2001). Competitive prices can also be used to measure benefits related to avoided costs associated with market goods and services. In analyzing benefits of the regulation, EPA used the avoided cost method to monetize three benefit categories: (1) maintenance of navigational waterways, (2) reservoir dredging, and (3) drinking water treatment. Other applicable market benefits (e.g., commercial fishing) are discussed qualitatively. The avoided cost approach represents an appropriate measure of social benefits for cases in which a policy change reduces costs to producers of a market-supplied good, but in which price effects are minimal (Boardman et al. 2001, p. 70-74).

5.1.2.1 Maintenance of Navigational Waterways

The primary effect of pollutant discharges from construction sites on navigation and transportation is accumulation of sediment in navigational channels and harbors (USEPA 2004b). *Section 2.4.1* of this report provides detail on sediment impacts on navigational waterways. Keeping these areas passable requires extensive dredging, which can be costly. Implementation of the regulation is expected to result in less frequent dredging of navigable waterways due to reduced sediment discharges from construction sites and, as a result, cost savings to the government and private entities responsible for maintenance of navigational shipping channels, harbors, and other waterways. EPA used the avoided cost of navigational waterways dredging as a measure of benefits resulting from the final regulation. *Chapter 7* of this report details methods and presents results of EPA's analysis of benefits to navigation expected from the regulation.

5.1.2.2 Water Supply and Use

- *Reservoir Dredging.* Water storage facilities (e.g., reservoirs) serve many functions, including providing drinking water, flood control, hydropower supply, and recreational opportunities. Sediment in streams can be carried into reservoirs, where it can settle and build up layers of silt over time. An increase in sedimentation rates will reduce reservoir capacity. *Section 2.4.2* of this report provides detail on sediment impacts on the reservoir water storage capacity. To replace this capacity, sediment must be dredged from reservoirs, or new reservoirs must be constructed (Clark et al. 1985). The regulation is expected to reduce the amount of sediment entering reservoirs and, as a result, the need for sediment mitigation measures in these reservoirs and the cost of reservoir maintenance. Because sediment mitigation is often accomplished through dredging, which itself is environmentally destructive, an additional benefit to reduced reservoir sediment buildup is the reduced environmental impact of dredging sediment and disposing of it. EPA used the avoided cost of reservoir dredging as a measure of benefits resulting from the final regulation. *Chapter 8* of this report details methods and presents results of EPA's analysis of benefits from reduced sedimentation of reservoirs due to the regulation.

- *Drinking Water Treatment and Household Water Use.* Discharges from construction sites can affect the quality and cost of providing drinking water. Sediment and other discharged pollutants negatively affect water quality, and require increased spending on treatment measures such as settlement ponds, filtration, and chemical treatment. There is an additional cost of removing sludge that is created during the treatment process (USEPA 2007b). EPA used the avoided cost of treating drinking water for sediment and disposing of sludge resulting from reduced turbidity in source water as a measure of benefits resulting from the final regulation. *Chapter 9* of this report details methods and presents results of EPA's analysis of benefits to drinking water treatment plants. Reducing nutrient loadings to surface waters is expected to reduce eutrophication which is one of the main causes of taste and odor impairment in drinking water. Taste and odor in drinking water has a major negative impact on the public perception of drinking water safety and the drinking water industry due to a significant increase in drinking water treatment costs from foul taste and odor in the source waters. The final regulation is expected to reduce the cost of drinking water treatment by improving taste and odor in the source waters. EPA, however, was unable to estimate a value of improved taste and odor of drinking water, due to data limitations (see *Chapter 9* for more detail).

- *Industrial Water Use.* Sediment may have negative effects on industrial water users. Suspended sediment increases the rate at which hydraulic equipment, pumps, and other equipment wear out, causing accelerated depreciation of capital equipment. Sediment can also clog cooling water systems at power plants and other large industrial facilities (Clark et al. 1985). *Section 2.4.4* of this report provides detail on sediment impacts on industrial water uses. The final regulation is expected to benefit industrial water users by reducing sediment concentrations in source waters and thus increasing the useful life of industrial equipment. The Agency, however, was not able to quantify this benefit category due to data limitations.
- *Agricultural Water Use.* Irrigation water that contains sediment or other pollutants from construction sites can harm crops and reduce agricultural productivity (Clark et al. 1985). *Section 2.4.5* of this report provides more detail on sediment impacts on agricultural water uses. The final regulation is expected to benefit agricultural producers by reducing sediment discharges and, as a result, sediment deposition on farm land; this would lead to improvements in land productivity and enhanced marketability of agricultural products. The Agency, however, was not able to quantify this benefit category due to data limitations.

5.1.2.3 Commercial Fishing

Sediment and other discharges from construction sites can greatly reduce fish populations by inhibiting reproduction and survival of an aquatic species. These population reductions would reduce the size of commercial harvest. These changes negatively affect subsistence anglers, commercial anglers and fish sellers, and consumers of fish and fish products. *Chapters 2* and 3 of this report provide some detail on aquatic life impacts from increased sediment and other pollutant concentrations in fresh and marine waters. Reducing sediment and other pollutant discharges from construction sites is expected to enhance aquatic life habitat and thus contribute to reproduction and survival of commercially harvested species, which in turn will lead to an increase in producer and consumer surplus. EPA did not quantify market benefits from improved commercial harvest resulting from this regulation due to data limitations. The *nonmarket* value of improved habitat for commercial fish species may be implicitly accounted for in the WTP for changes in environmental services provided by surface waters affected by construction site discharges stemming from the regulation (see *Chapter 10* of this report for details).

5.1.2.4 Public and Private Property Ownership

- *Property Values.* Aesthetic degradation of land and water resources resulting from sediment and other pollutant discharges (e.g., increased water turbidity) can reduce the market value of property and thus affect the financial status of property owners. For example, a hedonic price study by Bejranonda et al. (1999) found that "the rate of sediment inflow entering the lakes has a negative influence on lakeside property rent" (p. 216). Sediment discharges also have a significant impact on stream morphology.

For example, higher coarse sediment load leads to an increase in width of the river bed and, as a result, bank erosion (Wheeler et al. 2003). A 1993 study of Lake Erie's housing market found that "erosion-prone lakeshore property will be discounted" (Kriesel et al. 1993). Stabilization of stream banks leads to an increase in the value of surrounding property (Streiner and Loomis 1996). Therefore, the final regulation is expected to enhance nearby property value in two ways: (1) by improving the aesthetic quality of land and water resources (e.g., reduced turbidity) and (2) by improving stream morphology and stabilizing river banks.

Due to data limitations, EPA did not estimate changes in market values of properties located near water bodies expected to benefit from reduced sediment discharges from construction sites as a result of this regulation. The nonmarket component (i.e., increased satisfaction with the property) may be implicitly accounted for in WTP for improvements in environmental services provided by surface waters affected by construction site discharges (see *Chapter 10* of this report for details).

- *Flood Damages.* Sedimentation can also increase the severity of property damages from flooding. Sedimentation of river beds can reduce river capacity and result in higher flood levels and more frequent flooding. Additionally, sediments carried by flood waters can damage property and can be expensive to remove, particularly in developed areas. Clark et al. (1985) estimated flooding damages attributable to sediment discharges to be $1.5 billion (2008$), annually. *Section 2.4.6* of this report provides detail on sediment impacts on flood control and stormwater management. This regulation is expected to reduce flooding damages by reducing sediment discharges from construction sites. The Agency, however, was unable to estimate benefits from reduced flood damages due to data limitations.
- *Ditch Maintenance.* Sediment in discharges from construction sites can clog ditches and culverts. Such sedimentation can increase flooding, which may result in damages to bridges, roads, farmland, and other structures in the nearby areas. Preventing these damages may require increased maintenance efforts such as dredging (Clark et al. 1985). *Section 2.4.6* of this report provides more detail on sediment impacts on stormwater management. The final regulation is expected to reduce the costs of ditch maintenance by reducing the amount of sediment deposited in ditches. This benefit category is not quantified due to data limitations.

5.1.3 Nonmarket Benefits

EPA expects that this regulation will reduce the sediment and other pollutant discharges to surface waters and will enhance or protect aquatic ecosystems currently under stress, effects for which humans may hold value but cannot express this value in the form of a market transaction. The decrease in sediment and other pollutant loadings is expected to improve protection of resident species, enhance the general health of fish and invertebrate populations, increase their propagation to waters currently impaired, and expand fisheries for both commercial and recreational purposes or use values. Improvements in water quality such as decrease in turbidity will also favor increased recreational uses such as swimming, boating, and outings. Finally, the Agency expects that the regulation will augment nonuse values (e.g., option, existence, and bequest values) of the affected water resources.

The expected nonmarket benefits of water quality improvements fall into two broad categories: nonmarket use benefits and nonmarket nonuse benefits. The following sections describe these benefit categories and the nonmarket approaches to valuing these benefits in more detail.

5.1.3.1 Nonmarket Use Benefits

Direct use benefits include the value of improved environmental goods and services used and valued by people (even if these services and goods are not traded in markets). Aesthetic degradation of water resources resulting from sediment and turbidity discharges from construction sites can reduce owner satisfaction with the property and the residential area in general. It can also adversely affect recreational opportunities. Adverse impacts of sediment and other pollutant discharges and, as a result, increased turbidity of surface waters on recreational activities are summarized below.

- *Outings.* Activities that take place near water such as hiking, jogging, picnicking, and wildlife may be adversely affected by sediment and other pollutant discharges into the water. While these activities do not involve contact with the water, murky and visually unpleasant water and odors may greatly detract from the enjoyment gained through these activities. Decreases in fish populations may cause a reduction in wildlife near the resource, affecting wildlife viewing. Pollutants may negatively affect local flora and fauna, reducing the aesthetic appeal of the area near the resource and negatively impacting wildlife viewing.
- *Recreational Fishing*. As noted in *Chapter 2* of this report, sediment and other discharges from construction sites can reduce fish populations by inhibiting reproduction and survival of an aquatic species. This may lead to fewer and smaller fish, and a reduction of the game fish population. In addition, sediments and other pollutants reduce the aesthetics of the waterbody, which may reduce anglers' utility of their fishing experience. Additionally, turbidity caused by sediment and other pollutant discharges may affect recreational anglers by reducing the distance over which fish can see lures, resulting in lower catch rates (Clark et al. 1985).
- *Boating*. Polluted water greatly reduces the aesthetic appeal of recreational boating activities. Turbidity caused by sediment and other pollutant discharges may affect the safety of boating. Turbidity may obscure underwater obstacles, making collisions more likely. Increased sediment concentrations may create sandbars, increasing the chances of running aground. Clark et al. (1985) estimated that turbidity (from all sources) may be responsible for as many as 200 boating fatalities and many more injuries each year. Using the value for a statistical life's worth of risks ($7 million) and multiplying it by the number of boating fatalities yields the value of preventing these fatalities of $1.4 billion per year. Even if reducing discharges from construction sites prevents a few fatalities and injuries each year, the expected monetary value of preventing boating accidents can be significant.
- *Swimming.* Turbidity and other problems (e.g., eutrophication) associated with discharges from construction sites may greatly reduce a swimmer's aesthetic enjoyment of a resource. Additionally, turbidity may create safety hazards for the swimmer by reducing the ability to see underwater hazards or increasing diving accidents by impairing the ability to gauge water depth.
- *Hunting.* Similar to the effect on outings, discharged pollutants may greatly detract from the hunters' aesthetic enjoyment of a water resource. Damage to flora and fauna may also cause a reduction in the game population, reducing the number and quality of the game available.

Improved water quality from reducing sediment and other pollutant discharges from construction sites is expected to enhance the quality of living in the areas affected by construction activities, thereby resulting in welfare gain to the resident populations. The final regulation is also expected to enhance recreational uses of water resources affected by sediment discharges from construction sites, thereby resulting in welfare gain to recreational users of these resources. Improved water quality from reducing sediment and other pollutant discharges from construction sites may translate into two components of recreational benefits: (1) an increase in the value of a recreational trip resulting from a more enjoyable experience, and (2) an increase in recreational participation.

5.1.3.2 Nonmarket Nonuse Benefits

Even if no human activities (or uses) are affected by environmental changes caused by sediment and other pollutants from construction sites, such environmental changes may still affect social welfare. For a variety of reasons, including bequest, altruism, and existence motivations, individuals may value the knowledge that water quality is being maintained, that ecosystems are being protected, and that populations of individual species are healthy completely independent of their use value. It is often difficult to quantify the relationship between changes in pollutant discharges and the improvements in societal well-being that are not associated with current use of the affected ecosystem or habitat. That these values exist, however, is indisputable, as evidenced, for example, by society's willingness to contribute to organizations whose mission is to purchase and preserve lands or habitats to avert development (although some portion of these donations may be motivated by use values). Notwithstanding challenges involved in estimation of nonuse values, there is a substantial literature devoted to such issues (Bateman et al. 2002). This literature provides insight into analysts' ability to estimate nonuse values within various types of policy contexts, and for various types of resources.

5.1.3.3 Nonmarket Valuation Methods

It is frequently difficult to quantify and attach economic values to ecological benefits. The difficulty results from imperfect understanding of the relationship between changes in sediment, nutrient, and other pollutant discharges from construction sites and the specific ecological changes, ranging from a lack of water quality monitoring data for many locations, to time lags between water quality changes and changes in biological community condition. In addition, it may be difficult to attach monetary values to these ecological changes because they often do not occur in markets in which prices or costs are readily observed.

A variety of nonmarket valuation methods exist for estimating nonmarket use value, including both revealed and stated preference methods (Freeman 2003). Where appropriate data are available or may be collected, revealed preference methods can represent a preferred set of methods for estimating use values. These methods use observed behavior to infer users' value for environmental goods and services. Examples of revealed preference methods include travel cost, hedonic pricing, and random utility (or site choice) models. Compared to nonuse values, use values are often considered relatively easy to estimate, due to their relationship to observable behavior, the variety of revealed preference methods available, and public familiarity with the recreational services provided by surface water bodies.

In contrast to direct use values, nonuse values are often considered more difficult to estimate. Stated preference methods, or benefit transfer based on stated preference studies, are the generally accepted techniques for estimating these values (USEPA 2000b; OMB 2003). Stated preference methods rely on carefully designed surveys, which either (1) ask people about their WTP for particular ecological improvements, such as increased protection of aquatic species or habitats with particular attributes, or (2) ask people to choose between competing hypothetical "packages" of ecological improvements and household cost (Bateman et al. 2003). In either case, values are estimated by statistical analysis of survey responses.

Nonuse values may be more difficult to assess than use values for several reasons. First, nonuse values are not associated with easily observable behavior. Second, nonuse values may be held by both users and nonusers of a resource. Because nonusers may be less familiar with a resource, their values may be different from the nonuse values for users of the same resource. Third, the development of a defensible stated preference survey is often a time- and resource-intensive process. Fourth, even carefully designed surveys may be subject to certain biases associated with the hypothetical nature of survey responses (Mitchell and Carson 1989; Bateman et al. 2003). Finally, efforts to disaggregate total WTP into its use and nonuse components have proved troublesome (Carson et al. 2000).

To evaluate nonmarket benefits of this regulation, EPA developed a benefit transfer approach based on a meta-analysis of surface water valuation studies. Benefit transfer may be described as the “practice of taking and adapting value estimates from past research … and using them … to assess the value of a similar, but separate, change in a different resource” (Smith et al. 2002, p. 134). It involves adapting research conducted for another purpose to estimate values within a particular policy context (Bergstrom and De Civita 1999). Although the use of primary research to estimate values is generally preferred, the realities of the policy process often dictate that benefit transfer is the only option for assessing certain types of nonmarket values (Rosenberger and Johnston 2007). In the benefit transfer used for analyzing benefits of the regulation, the meta-analysis is of WTP that incorporates both use and nonuse values.

Although the potential limitations and challenges of benefit transfer are well established (Desvousges et al. 1998), the Agency also emphasizes that benefit transfers are a nearly universal component of benefit cost analyses conducted by and for government agencies. As noted by Smith et al. (2002, p. 134), “nearly all benefit cost analyses rely on benefit transfers, whether they acknowledge it or not.” Benefits transfer methods may be placed into three general categories: (1) transfer of an unadjusted fixed-value estimate generated from a single study; (2) the use of expert judgment to aggregate or otherwise alter benefits to be transferred from a site or set of sites, and (3) estimation of a value estimator model or benefits transfer function, often based on data gathered from multiple sites (Bergstrom and De Civita 1999).

Given the generally unreliable performance of unadjusted single-site transfers, EPA used meta-analysis of 45 surface water valuation studies that use stated preference techniques to estimate a benefit function that allows forecasting of total WTP estimates (including use and nonuse values) in a variety of policy contexts, thereby providing more valid benefit estimates for transfer applications (USEPA 2000b; Johnston et al. 2005). Because stated preference surveys describe a range of environmental services corresponding to different levels of water quality (e.g., suitable for boating or swimmable), the total WTP derived from meta-analysis implicitly accounts for a wide range of nonmarket use and nonuse values, including recreation, aesthetic value of near-water properties, enhanced quality of source water for drinking and household use, as well as existence and bequest values.

The technical details involved in the estimation of original meta-analyses are presented in *Appendix G* as well as in sources such as Bateman and Jones (2003), USEPA (2004d), Johnston et al. (2005, 2006), Rosenberger and Phipps (2007), and Shrestha et al. (2007). *Chapter 10* of this report describes application of the meta-analysis results for estimating benefits of water quality improvements resulting from the regulation.

5.2 Summary of Effects

Ultimately, changes in sediment and other pollutant loadings from construction sites related to the final regulation affect humans through influences on the production of goods and services that enter into individuals’ utility functions. *Table 5-1* summarizes some of the important ways in which sediment and other pollutant discharges from construction sites affect the quantity and quality of services provided by the natural environment and how these services in turn provide economic value to society through various use and nonuse related societal welfare mechanisms.[3] EPA was not able to bring the same depth of analysis to all of the environmental services affected by the regulation categories, however, because of imperfect understanding of the link between discharge reductions and benefit categories, and how society values some of the benefit events. EPA was able to quantify and monetize some benefits (including maintenance to navigable waterways, reservoir dredging, drinking water treatment, and WTP for improvements in large river and streams and some coastal waters), quantify but not monetize other benefits (changes in pollutant concentrations in lakes), and assess still other benefits only qualitatively (e.g., commercial fishing, flood damages, and property values).

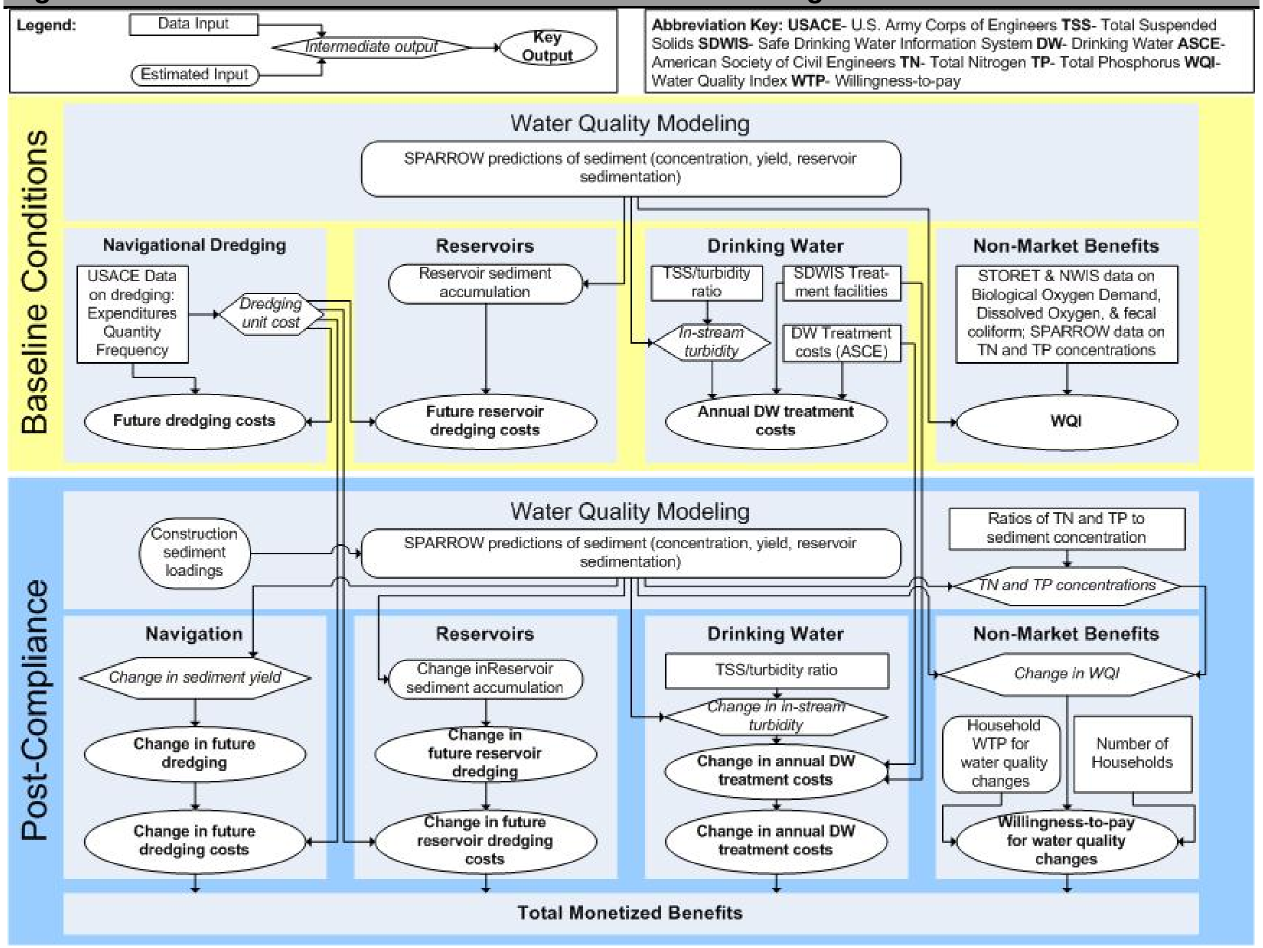
[3] Underlying this table is a conceptual framework based on the valuation of ecosystem services (Daily 1997; Simpson and Christensen 1997; van Houtven et al. 2005).

Table 5-1: Summary of Benefits from Reducing Sediment and Other Pollutant Discharges from Construction Sites

Activity	Environmental Service Flows Potentially Affected by Discharges from Construction Sites	Entities Affected by Environmental Changes			Methods Used in EPA's Analysis of Benefits from the Regulation
		Firms	Governments	Individuals	
Recreation ➢ Outings ➢ Boating ➢ Swimming ➢ Fishing	Aesthetics, water clarity, water safety, degree of sedimentation, eutrophication, weed growth, fish and shellfish populations	✓		Use & Nonuse Values	Benefits Transfer
Commercial Fishing and Shellfishing	Fish and shellfish populations	✓	✓	Use & Nonuse Values	Qualitative Discussion
Property Ownership	Aesthetics, safety of property from flooding, property value	✓	✓	Use & Nonuse Values	Benefits Transfer
Water Conveyance and Supply ➢ Water conveyance ➢ Water storage ➢ Water treatment	Turbidity, degree of sedimentation	✓	✓	Use Values	Avoided cost to governments, including: ➢ Reservoir dredging ➢ Drinking water treatment
Water Transportation	Degree of sedimentation of navigational channels, harbors, and other waterways	✓	✓	Use Values	Avoided cost to firms and governments
Water Use ➢ Industrial ➢ Municipal ➢ Agricultural	Turbidity	✓	✓	Use Values	Qualitative Discussion
Knowledge (No Direct Uses)	Environmental health			Nonuse	Benefits Transfer

Figure 5-1 depicts data sources, analytic steps, and models in EPA's analysis of benefits from reducing discharges of sediments and nutrients from construction sites. *Chapters 7* through *11* detail methods used in these analyses and present results.

Figure 5-1: Calculation of Monetized Benefits from the Regulation



6 Water Quality Modeling

Changes in water quality resulting from the various regulatory options were assessed using the U.S. Geological Survey's SPARROW models (**SPA**tially **R**eferenced **R**egressions **O**n **W**atershed attributes). SPARROW models for the coterminous United States for sediment, nitrogen, and phosphorus were used to characterize baseline water quality conditions. For each regulatory option, reductions in sediment discharge from constructions sites were predicted as described in *Development Document for Final Effluent Guidelines and Standards for the Construction and Development Category* (USEPA 2009b). These reductions were modeled in SPARROW and resulting changes in in-stream sediments and nutrient concentrations were evaluated to assess the environment impact of each option.

EPA did not calculate reductions in discharge levels of other pollutants (e.g., turbidity, metals, pesticides) under each of the alternative regulatory options due to insufficient data on the frequency and magnitude of their presence in construction site discharges in the United States. It was therefore not possible to quantify water quality improvements associated with reduction of their discharge. These pollutants are instead discussed qualitatively in *Chapters 2 and 3*. Some of these pollutants erode and travel with sediment discharges and are therefore addressed to some degree by the same practices that reduce sediment discharges.

This chapter describes the methodology used to estimate the water quality impacts of the alternative regulatory options and is organized as follows: *Section 6.1* describes the general SPARROW modeling approach. *Section 6.2* provides details on the SPARROW sediment model. *Section 6.3* describes the approach used to estimate water quality impacts in estuaries where no model flow predictions were available from SPARROW. *Section 6.4* describes the approach for modeling nutrient reductions associated with reduced discharges. *Section 6.5* provides a brief summary of predicted sediment loading reductions that are used in the model. *Section 6.6* discusses model results.

6.1 SPARROW Model Documentation[4]

6.1.1 General Overview

SPARROW is a watershed modeling technique for estimating contaminant source contributions and transport in surface waters. SPARROW employs a statistically estimated nonlinear regression model with contaminant supply and process components, including surface-water flow paths, non-conservative transport processes, and mass-balance constraints. Regression equation parameters are estimated by correlating stream water-quality records with GIS (Geographic Information System) data on pollutant sources (point and nonpoint) and climatic and hydrogeologic properties (e.g., precipitation, topography, vegetation, soils, and water routing) that affect contaminant transport. The SPARROW parameter-estimating procedure also provides measures of uncertainty in water-quality predictions.

SPARROW model infrastructure consists of a detailed watershed – stream reach network to which all monitoring stations and watershed properties are spatially referenced. Digital elevation models (DEMs) are used to delineate watershed boundaries and to identify overland flowpaths. The spatially distributed model structure allows separate statistical estimation of land- and water-related rates of pollutant delivery

4 Discussion adapted from Schwarz et al. (2006).

from sources to streams, and transport of pollutants to downstream locations within the stream network (reaches, reservoirs and estuaries). Mechanistic separation of terrestrial and aquatic features of large watersheds and improved parameter estimation techniques represent significant advances in water-quality modeling of important contaminant sources and watershed properties that control transport over large spatial scales. SPARROW has been applied in the analysis of suspended sediment, surface-water nitrogen and phosphorus nutrients, pesticides, organic carbon, and fecal bacteria, and is potentially applicable to other measures of water quality. The SPARROW software is written in Statistical Analysis System Interactive Matrix Language (SAS IML).

The remainder of this chapter provides a brief description of SPARROW modeling concepts, model specification, data sources, and estimation results. *Appendix C* of this report provides more detail on the nature of the general SPARROW methodology. *Appendix D* provides more detail on the suspended sediment SPARROW model developed by USGS (Schwarz 2008a). *Appendix C* also provides more detail on the total nitrogen (TN) and total phosphorus (TP) SPARROW models developed by USGS (Smith et al. 1997).

6.1.2 Modeling Concept

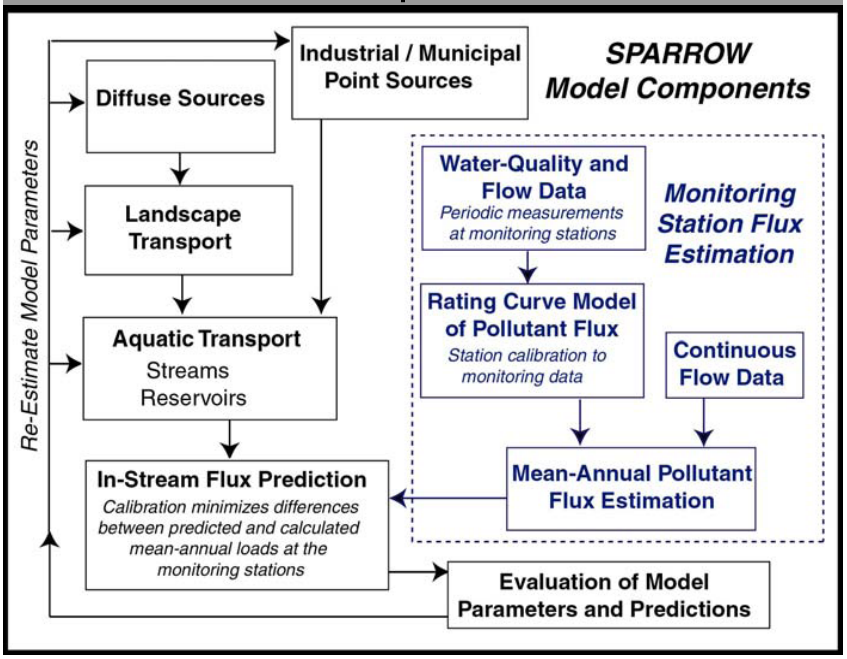
The broad objective of SPARROW modeling is to establish a mathematical relationship between water-quality measurements made at a network of monitoring stations and attributes of the contributing watersheds. The model may be used to satisfy a variety of water-quality information objectives. One objective is to describe past or present water-quality conditions for a state or region on the basis of monitoring data. A second objective is to identify and quantify the sources of pollution that give rise to in-stream water-quality conditions. A third objective is water quality simulation: the ability to portray counterfactual conditions for specified inputs. Water-quality simulations can depict the in-stream effects of changes in contaminant sources associated with alternative potential pollution-control strategies, providing an important step in the analysis of benefits associated with specific Best Management Practices (BMP) or other regulatory approaches. For the analysis described in this document, SPARROW is used to evaluate the effectiveness of a range of regulatory options for controlling construction related runoff and improving water quality as measured by in-stream Total Suspended Solids (TSS), TN and TP concentrations. A fourth objective of SPARROW modeling is to test one or more hypotheses about the nature and importance of factors and processes that may have influenced water quality at the locations where samples were collected.

The dependent variable in SPARROW models is the mass flux, the mass of contaminant that passes a specific stream location per unit time, and the laws of conservation of mass apply. Mass accounting in SPARROW models is supported by the explicit spatial structure defined by the stream network. The imposition of mass balance greatly improves the interpretability of model coefficients. For example, assuming mass balance, the coefficient associated with the reach time-of-travel variable is interpreted as a first-order decay rate. Mass balance provides a basis for contaminant mass flux accounting, whereby mass flux can be allocated to its various sources, both spatially and topically. Spatial variability is modeled as a function of natural and human-related properties of watersheds that influence the supply and transport of contaminants. Estimates of long-term mean mass flux are developed from water-quality and streamflow monitoring data that are regularly collected at fixed locations on streams and rivers.

6.1.3 Model Infrastructure

A flow diagram is provided in *Figure 6-1* to illustrate the functional linkages between the major spatial components of SPARROW models. Mass flux estimates at monitoring stations are derived from station-specific models that relate contaminant concentrations from individual water-quality samples to continuous records of streamflow and time. The *stream reach*, inclusive of its incremental contributing drainage basin, is the most elemental spatial unit of the SPARROW model infrastructure. Explanatory data (e.g., climate, topography, land use) are frequently compiled according to geographic units that are not coincident with the drainage basin boundaries of river reaches. These data may be collected at different spatial scales and according to spatial units that reflect political boundaries (e.g., counties) or other non-hydrologic features of the landscape.

Figure 6-1: Schematic of the Major SPARROW Model Components



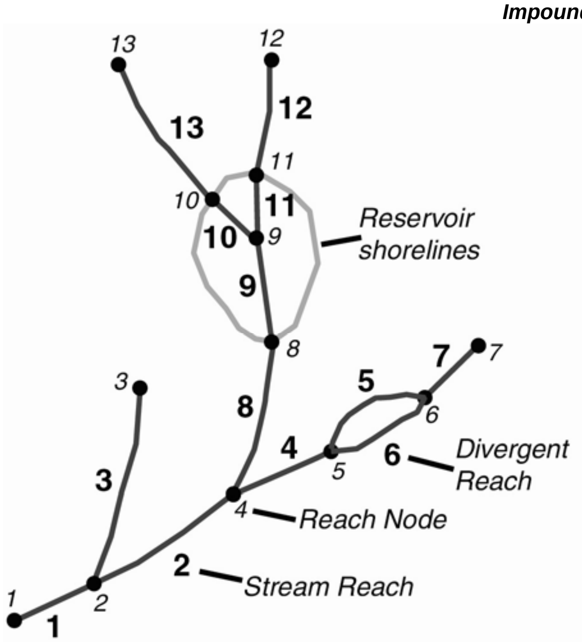
Source: Modified from Alexander et al. (2002a).

The estimation of a SPARROW model requires estimates of long-term mean mass flux from a spatially distributed set of monitoring stations, having sufficiently long periods of record, inclusive of a wide range of spatial scales and expressing considerable variation in predictor variable conditions. Long-term mean mass flux is obtained by relating infrequently collected water-quality samples to continuous measurements of streamflow and functions of time. In order to have consistency across stations having different periods of record, it is necessary to detrend the mass flux estimates to a common base year prior to computing the long-term mean.

A vector- or raster-based digital representation of the stream and river network topology is the most fundamental component of the spatial infrastructure that supports the SPARROW model *(Figure 6-2)*. The node topology is defined according to an upstream (from-) and downstream (to-) node attribute table.

Figure 6-2: Schematic Illustrating a Vector Stream Reach Network with Node Topology and Water/Contaminant Reach-Node Routing Table

The reach-type indicator has possible values of "0" (Stream Reach), "1"(Impoundment Reach), and "2" Outlet Reach for Impoundment.



Reach-Node Attribute Table

Reach Number	Up-stream Node	Down-stream Node	Diversion Fraction	Reach-Type Indicator
13	13	10	1.0	0
10	10	9	1.0	1
12	12	11	1.0	0
11	11	9	1.0	1
9	9	8	1.0	2
8	8	4	1.0	0
7	7	6	1.0	0
5	6	5	0.7	0
6	6	5	0.3	0
4	5	4	1.0	0
2	4	2	1.0	0
3	3	2	1.0	0
1	2	1	1.0	0

SPARROW separately estimates the sediment and contaminant attenuation in reservoirs and lakes. The assessment of pollutant mass flux to coastal estuaries requires an expanded reach network that includes shoreline features and the identification of reaches that terminate at estuaries. Shoreline reaches are used to define coastal drainage areas – areas that discharge runoff directly to the estuary without transport through a stream reach. The SPARROW node and routing architecture also can fully support the modeling of contaminant transport along "off-reach" (landscape) flow paths according to flow directions defined by landscape topography as reflected, for example, in DEMs.

The explanatory variables evaluated in SPARROW reflect current knowledge of natural and human-related sources and the physical, chemical, or biological properties of the terrestrial and aquatic ecosystems that affect supply and transport of contaminants in watersheds. Point- and nonpoint-source variables may include direct measures of the supply of contaminant mass to the landscape and streams and reservoirs (e.g., municipal and industrial wastewater discharge, fertilizer application), or they may serve as surrogate indicators of the contaminant mass supplied by point and diffuse sources in watersheds, such as land-use/land-cover data or census data on human and livestock populations. Climatic and landscape properties that affect contaminant transport may include measures of water-balance terms (e.g., solar radiation, precipitation, evaporation, evapotranspiration), soil characteristics (e.g., permeability, moisture content), water-flow path properties (e.g., slope, hydraulic roughness, topographic index), or management practices and activities (e.g., tile drains, conservation tillage, BMPs).

6.1.4 Model Specification

The specification of a SPARROW model consists of identifying the explanatory variables and functional forms. Conceptually, the contaminant mass flux leaving a reach is the sum of two components. The first component is the mass flux generated within upstream reaches that is delivered to the reach via the stream network. Losses of contaminant mass from the stream network may occur at points where flow is diverted, and contaminant mass will generally be attenuated by stream or reservoir processes. The second component consists of source loading that is generated within the reach's incremental watershed and delivered to the stream network within the reach. A number of source-dependent processes affect the amount of source loading reaching the stream network and transported to the reach's downstream outlet node. The processes affecting delivery to the stream network are called land-to-water delivery processes, and may include both surface and sub-surface elements. Land-to-water delivery processes determine the amount of contaminant generated within an incremental drainage area delivered to the area's corresponding reach.

Large digital spatial data sets are now readily available, and advances in digital topographic and stream network data [e.g., National Hydrography Dataset (NHD), National Land Cover Database (NLCD)] have made spatially distributed modeling more feasible. Inclusion in the SPARROW model of detailed information on the geographic locations where contaminants are released in watersheds has particular relevance to the policy and management applications of the model. The source terms used in the model can be generally classified as *intensive* and *extensive* measures of contaminant mass. *Intensive* measures are direct measures of pollutant mass, such as fertilizer application, livestock waste, atmospheric deposition, or sewage-effluent mass flux. The associated source parameter is a dimensionless coefficient that, together with standardized expressions of the land-to-water delivery factor, describes the proportion or fraction of the source input that is delivered to streams. *Extensive* measures of contaminant mass are surrogate indicators and include measures of watershed properties such as specific land-use area and sewered population, considered to be proportional to the actual contaminant mass generated by a particular type of source. The associated model coefficients are expressed as the contaminant mass generated per unit of the source type (e.g., kilograms kilometer^{-2} year^{-1}; kilograms person^{-1} year^{-1}). Combined with the land-to-water delivery factor, the standardized source coefficient indicates the mean quantity of contaminant mass per unit of the surrogate source measure that is delivered to streams. For land-use terms, the standardized coefficient gives what is frequently cited as an export coefficient.

Landscape variables in SPARROW describe properties of the landscape that relate to climatic, natural- or human-related terrestrial processes affecting contaminant transport. The model structure allows tests of hypotheses about the influence of specific features of the landscape on contaminant transport. Landscape variables may include water-balance terms (e.g., precipitation, evapotranspiration) related to climate and vegetation, soil properties (e.g., organic content, permeability, moisture content), topographic water flow-paths variables (e.g., overland flow, topographic index, and slope), or management practices and activities, including tile drainage, conservation tillage practices, and BMPs related to stream riparian properties. Particular types of land-use classes, such as wetlands or impervious cover, may also be potentially used to describe transport properties of the landscape.

Stream attenuation processes that act on contaminant mass as it travels along stream reaches are frequently modeled as first-order reaction rate processes. A first-order decay process implies that the rate of removal of the contaminant from the water column per unit of time is proportional to the concentration or mass that is present in a given volume of water (a zero-order process corresponds to a constant rate of removal per unit of time). Accordingly, in basic forms of the SPARROW model, the fraction of the contaminant mass originating from the upstream node and transported along a reach to its downstream node is estimated as a function of the mean water time of travel. Attenuation processes that act on contaminant mass as it travels through a lake or reservoir are often modeled as a net removal process, with the loss coefficient expressed as either a first-order *reaction rate* or a *mass-transfer coefficient* (also referred to as an *apparent settling velocity*). These mass-balance models typically assume steady-state and uniformly mixed conditions in the waterbody.

6.1.5 Model Estimation

The SPARROW model equation is a nonlinear function of its parameters, and the model must be estimated using nonlinear techniques. SPARROW utilizes a nonlinear weighted least squares (NWLS) estimation method, implemented in SAS. Model parameters are evaluated for statistical significance and the quantification of uncertainty. The key parameter statistics include the estimated mean coefficient values, estimated variance of these coefficient estimators, and measures of statistical significance based on the *t* statistics. Parameters are also evaluated for physical interpretability, to test hypotheses about the importance of different contaminant sources and the hydrologic and biogeochemical processes that are represented in the model. The interpretability of the parameters and their relation to specific processes are enhanced in SPARROW by the use of a mass balance, mechanistic structure that explicitly separates the terrestrial and aquatic properties of watersheds and accounts for nonlinear interactions among watershed properties. The sign of SPARROW model coefficients can be evaluated to determine the direction of influence of any explanatory variable on in-stream estimates of mean-annual mass flux. Coefficients associated with source inputs expressed in areal units describe the mass per unit area delivered to streams from these land areas. Other source coefficients that are expressed in dimensionless units provide a measure of the fraction of the contaminant that is delivered from each source to streams, rivers, and reservoirs.

6.2 SPARROW Sediment Model[5]

The following describes results from a SPARROW suspended sediment model developed for the coterminous United States and described in Schwarz (2008a), which is reprinted in full in *Appendix D* of this document. The analysis is based on mass flux estimates compiled from more than 1,800 long-term monitoring stations operated by the U.S. Geological Survey (USGS) during the period 1975-2007. The SPARROW model is structured on the Reach File 1 (RF1) stream network, consisting of approximately 62,000 reach segments. The reach network has been modified to include more than 4,000 reservoirs, an important landscape feature affecting the delivery of suspended sediment. The model identifies six sources of sediment, including the stream channel and five classes of land use (urban, forested, federal non-forested, agricultural and other). The delivery of sediment from landform sources to RF1 streams is mediated by soil permeability, erodibility, slope, and rainfall; streamflow is found to affect the amount of sediment mobilized from the stream channel. The results show agricultural land and the stream channel to be major sources of sediment. Per unit area, federal non-forested and urban lands are the largest landform sediment sources. Reservoirs are identified as major sites for sediment attenuation.

6.2.1 Sediment Model Data Sources

The spatial framework of the SPARROW sediment model is the vector-based 1:500,000 scale River RF1 hydrography, originally developed by EPA (USEPA 1996b) and enhanced to include areal hydraulic load information for selected reservoirs, shoreline reaches, and reach catchment areas derived from the USGS Hydro 1K DEM (USGS 2006a). The enhanced network, consisting of 62,776 reach segments, including shoreline reaches, 61,214 delineated reach catchments, and 2,171 individual reservoirs, has been used to support numerous national SPARROW modeling efforts for the coterminous United States. The RF1 reach network for the current SPARROW sediment model was further enhanced by inclusion of areal hydraulic load information for approximately 2,000 additional large reservoirs from the National

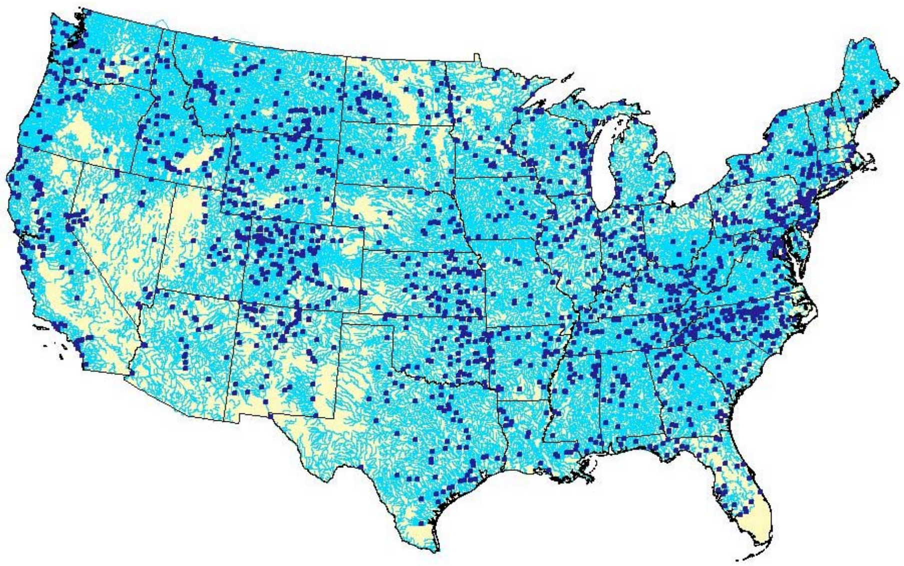
[5] Adapted from Schwarz (2008a).

Inventory of Dams (NID, USACE 1996). The dependent variable in the SPARROW sediment model is given by long-term mean sediment mass flux, detrended to the base year 1992. In-stream sediment concentrations and stream discharge measurements over the water year (WY) period 1975-2007, have been obtained from the National Stream Quality Accounting Network (USGS 2006b), the USGS NAWQA Program (USGS 2001), and the USGS National Water Information System (NWIS; USGS 2008), a database encompassing USGS water quality monitoring stations and water quality monitoring activities done in cooperation with State governments. The sediment model is estimated using 1,828 monitoring stations on the RF1 stream network *(Figure 6-3)*.

While the RF1 network provides reasonably comprehensive national coverage of major rivers, streams and other surface water bodies, coverage is limited in certain important respects. First, RF1 network coverage is limited to the coterminous United States, thus excluding Alaska and Hawaii. In addition, while RF1 1:500,000-scale network reaches have associated data or estimates of stream discharge and velocity that are required to specify the SPARROW model, the network excludes the majority of the nation's total stream mileage, and smaller streams in particular. The linear coverage of the RF1 network is approximately 700,000 miles (USEPA 2007e). By contrast, coverage of the USGS National Hydrographic Dataset (NHD) at 1:24,000 - 1:100,000 scale is currently over 7 million miles (USGS 2007b). Non-homogeneity of coverage may lead to an under-estimation of the impacts of construction-related sediment on smaller stream reaches. As construction activities may be concentrated along lower-order streams not included in the RF1 network, the relative share of total sediments contributed by construction activities in these reaches may be high during active construction. By contrast, the specific impacts of construction activities may diminish in magnitude relative to contributions from spatially extensive and diffuse land uses, including agriculture, at the spatial level of RF1 reaches.

Data on land cover and land use have been developed from the 2001 USGS National Land Cover Data Set Retrofit Change Product (Multi-Resolution Land Characteristics Consortium (MRLC) 2001), derived from Landsat TM/ETM remotely sensed imagery at 30-meter resolution and classified according to the eight Anderson Level I categories. For use within SPARROW, data were transformed to 1 km^2 cells within a Lambert map projection as consistent with Hydro 1K. The 1 km^2 cells were then resolved to catchments associated with specific RF1 reaches. For model estimation, land use was given by the 1992 values of the 2001 NLCD Retrofit Change Product. The 2001 values of the Retrofit Change Product were used to simulate water quality conditions for 2001. Federal range and barren land was included in the model separately from private land. Federal land extent, taken from the Federal Land coverage of the National Atlas (USGS 2003) and transformed to 1 km^2 cells, was apportioned using the 1 km^2 transformation of the 1992 NLCD Change Product for Anderson Level 1 range and barren land classes. For model simulation of 2001 conditions, federal range and barren land were similarly estimated using the land use estimates from the 2001 NLCD Retrofit Change Product.

Figure 6-3: Location of 1,828 Water Quality Monitoring Stations Used in the SPARROW Sediment Model, in Relation to the Reach File 1 (RF1) Reach Network



Variables determining the delivery of contaminants from the land to RF1 streams include soil erodibility (RUSLE k-factor), soil permeability, mean slope, and precipitation (RUSLE r-factor). Slope, soil erodibility, and permeability were obtained from the State Soil Geographic (STATSGO) database (USDA 1994), converted to a 1-km^2 grid and averaged over RF1 catchments. The RUSLE rainfall factor was derived by interpolating a digitized national map of rainfall factor isoline contours, creating a continuous 1 km^2 grid surface, which was subsequently averaged over individual RF1 catchments.

6.2.2 Model Estimation Results

The nonlinear least squares estimation results of a preliminary version of the SPARROW suspended sediment model are given in *Table 6-1.* The model includes six source terms: five measured by area of specific land use (expressed in km^2), and the sixth by the length of stream channel. The five land use sources are urban land, forested land, federal nonforested land, agricultural land, and other land. The federal land class excludes federal forested land, which is incorporated in the forested land class. Agricultural land includes cropland, pasture land and orchards. Other land consists of nonfederal range and barren land. Among all the land classes, only wetlands and land covered by water, ice or snow are excluded as a potential source. The source described as “streambed” relates to stream channels as a direct source of sediment, and is measured in terms of stream length (expressed in meters). The model specifies two in-stream sediment attenuation processes: attenuation in streams, distinguished by three streamflow classes (below 500 cubic feet per second (cfs), 500–1,000 cfs, and greater than 1,000 cfs); and reservoir attenuation, specified as a function of areal hydraulic load.

Table 6-1: Estimation Results for the SPARROW Suspended Sediment Model

Parameter	Units	Estimate	Standard Error	p-value
Source Coefficients				
Urban land	$kg/km^2/yr$	47,130	9,925	0.000
Forested land	$kg/km^2/yr$	634	898	0.480
Federal nonforested land	$kg/km^2/yr$	64,344	12,411	0.000
Agricultural land	$kg/km^2/yr$	18,047	3,623	0.000
Other land	$kg/km^2/yr$	11,343	3,186	0.000
Streambed (reach length)	kg/m/yr	28.80	6.40	0.000
Land to Water Delivery Factors				
Slope	–	0.804	0.087	0.000
Soil permeability	–	-0.778	0.094	0.000
R-factor	–	0.821	0.081	0.000
K-factor	–	1.292	0.279	0.000
Flow [< 500 cfs] (Reach)	–	0.154	0.100	0.125
Flow [> 500 cfs] (Reach)	–	0.721	0.354	0.042
Stream Attenuation Factors				
Travel time (Q < 500 cfs)	day^{-1}	-0.007	0.016	0.673
Travel time (500 < Q < 1,000 cfs)	day^{-1}	-0.233	0.057	0.000
Travel time (Q > 1,000 cfs)	day^{-1}	0.009	0.047	0.854
Reservoir settling velocity	m/yr	36.49	5.552	0.000
Number of Observations	1,828			
RMSE	1.414			
R-square	0.711			

Source: Schwarz (2008a).

With the exception of forested land, all of the source variables are highly statistically significant. The largest intrinsic sediment yield is associated with federal range and barren land; urban land has the second highest intrinsic yield. Stream channels are also a statistically significant source of sediment. Land-to-water delivery for land sources is strongly mediated by the four delivery variables: soil permeability, soil erodibility, slope, and rainfall. With the exception of soil permeability, an increase in these factors results in an increase in the share of sediment delivered to streams. Permeable soils reduce the delivery of sediment, presumably because more runoff infiltrates, leaving less overland flow to carry sediment. Greater streamflow causes an increase in the amount of sediment generated from stream channels, with the largest effect associated with streams with flow greater than 500 cfs. Reservoir retention is statistically significant and indicates sediment settles at a mean velocity of 36 meters per year. The preliminary model does not find evidence of sediment attenuation in streams, as in-stream attenuation in small and large streams is not statistically different from zero. Additional investigation would be necessary to determine if this result is real, or if there are additional reach attributes, currently absent from the model, that identify a subset of reaches where sediment attenuation takes place.

6.2.3 Model Simulation

The estimated SPARROW suspended sediment model for base water year 1992 is used to simulate water-quality conditions for 2001, with and without proposed changes in the regulation of construction activity. The simulation of suspended sediment mass flux for water year 2001 without changes in regulation is obtained with all land use-related source variables set to 2001 values based on the NLCD Retrofit Change Product. Flow weighted concentration is estimated by dividing simulated mass flux estimates by mean streamflow over the period WY 1975-2006. Although the SPARROW model does not explicitly include a term for construction loading, such loading is implicitly accounted for in the urban land component of the model. Therefore, the 2001 pre-compliance loading from construction (the “base-case” scenario loading) is incorporated in the 2001 mass flux attributed to urban land that is obtained by evaluating the urban land variable in the SPARROW model using the 2001 NLCD Change Product value.

The absence in the SPARROW model of an explicit term for construction loading requires the development of an indirect method for assessing changes in sediment mass flux arising from different construction industry regulation scenarios. Suspended sediment loading under alternative regulation scenarios has been estimated by EPA using a variation of the USLE. The USLE method determines the amount of soil that is mobilized and delivered, under a regulatory scenario, to the edge of a construction site. To evaluate the impact that changes in these loadings have on RF1 stream sediment mass flux and concentration, it is first necessary to assess the rate at which "edge of site" loads are subsequently delivered to RF1 streams. We use the estimated rate of delivery from agricultural land (explicitly included in the model) that can be factored into a mobilization ("edge of site") delivery component and a stream-delivery component. The method uses edge-of-site measurements of agricultural sediment erosion from the 1992 National Resources Inventory (NRI) to isolate the stream-delivery component from the overall rate of delivery from agricultural land and applies this component to the change in construction loading to determine the change in loading to RF1 streams. Thus, the approach assumes that the delivery of sediment from the edge of a site to an RF1 stream is the same for both construction and agriculture activities; although the mobilization and delivery of sediment to the edge of the site between these activities is allowed to differ.

6.3 Extension of SPARROW to Estuaries and Coastal Waters

EPA used SPARROW output to evaluate sediment deliveries to estuaries. While SPARROW predicts mass flux in all reaches, it does not estimate concentrations in reaches where flow estimates are not available. Of the 2,067 estuarine/coastal reaches in the RF1 dataset, SPARROW provided TSS concentrations in more than 92 percent of the reaches. Of those reaches, 772 are retained for water quality analysis based on the availability of data for other parameters needed in calculating Water Quality Index values (see *Chapter 10).* Of the 772 reaches used in the analysis, SPARROW provides TSS concentrations for 729 reaches in this set directly. In the remaining 43 estuarine reaches, sediment concentrations were calculated using the SPARROW predicted mass flux and dissolved concentration potentials (DCPs) for estuaries (USEPA 1997). The National Oceanic and Atmospheric Administration (NOAA) developed DCPs to provide an approach for predicting the ambient concentration of conservative contaminants that are introduced into an estuary and are then subject to mixing and dilution. Using this approach, EPA was able to estimate TSS concentrations associated with annual quantities of sediment discharged into estuaries as predicted by SPARROW without detailed hydrodynamic modeling of estuaries.

Concentration calculations using DCPs presume that water quality constituents are dissolved, conservative pollutants dispersed under well-mixed, steady-state conditions. These criteria are not strictly met for TSS, although the finest particle fractions approximately meet these criteria due to their extremely low settling velocities and relatively stable composition. Because current and proposed sediment abatement practices remove coarser particle fractions first, construction site sediment discharges are expected to contain primarily finer particles. Therefore, the DCP methodology criteria are assumed to be applicable for estimating changes in estuary water quality associated with changes in construction site sediment discharge.

NOAA has calculated DCP values for most major estuaries in the Southeast (NOAA and EPA 1989a); Northeast (NOAA and EPA 1989b); Gulf of Mexico (NOAA and EPA 1989c); and West Coast (NOAA and EPA 1991) of the coterminous United States. For these estuaries, DCP values and baseline TSS concentrations calculated using DCPs are shown in *Appendix E* in *Table E-1* through *Table E-4* for Southeastern, Gulf, Northeastern, and West Coast estuaries, respectively. Data on these and additional estuaries are found via the NOAA Coastal Geospatial Data Project (http://coastalgeospatial.noaa.gov).

Total sediment loadings estimated in SPARROW for reaches draining to estuaries were aggregated on the basis of 5-digit NOAA Estuarine Drainage Area (EDA) codes. Estimated loadings were calculated in SPARROW as Suspended Sediment Concentrations (SSC). Suspended Sediment Concentrations needed to be adjusted before the application of DCP methods, for two reasons. First, reported SSC values include all particle size fractions and coarse sediment fractions are not assumed to behave as conserved within the water column due to higher settling velocities. Approximately 127,000 sediment records for which particle size distributions were recorded, collected at 1805 water quality gauging stations obtained from USGS (Schwarz 2008b) were analyzed. It was found that on average over these samples, 78.4 percent of suspended sediment mass was classified as “fine” (diameter < 0.0625 mm). This percentage varied to some extent by 2-digit USGS Water Resources Regions, from 71 percent (Region 14) to 89.5 percent (Region 9); and was subject to even wider at-site inter-seasonal and inter-annual variations. It is assumed that the implied sediment size distribution of SPARROW output is equivalent to observed distributions. However, only 15 of the 1,805 water quality gauging stations for which sediment size fraction information was available were terminal or coastal reaches (draining directly into bays or estuaries), so that no regional or estuary-specific sediment size distributions could be identified.

A second consideration is that USGS (2000) has determined that SSC and TSS, while carrying the same physical interpretation, are subject to systematic differences in measurement due to differing laboratory protocols. SSC values in the SPARROW output are divided by 1.3 to adjust for this difference, as described in *Section 6.6.* The use of this adjustment factor yields TSS concentrations that are roughly 77 percent of corresponding SSC values. Since this adjustment is reasonably close to the nation-wide (average of samples) percentage of fine particles in SSC (78 percent), SPARROW SSC values were converted to TSS equivalent, and the TSS values assumed to conform approximately to the assumptions underlying DCP calculations. In order to determine how well these assumptions are reflected in observed estuarine water quality conditions, data on suspended sediment size distribution for 623 samples taken within San Francisco Bay were obtained from the USGS (http://sfbay.wr.usgs.gov/access/wqdata/). The percentage of fine particles (< 0.0625 mm) was 71 percent for these samples. It is evident that suspended sediment accounting within estuaries is complex, and ambient TSS levels, in addition to the percentage of fine particles, are observed to reflect many influences beyond the annual riverine input of sediment. As an additional check on SPARROW outputs to estuaries, the aggregated annual loadings per unit of contributing watershed area were calculated for each channel contributing sediment to estuaries, and found to be reasonable in most cases.

Annual sediment loads as estimated by SPARROW were aggregated by estuary, unit-converted and divided by 10,000 tons per year (the DCP unit loading), and multiplied by DCP coefficients to obtain ambient estuarine TSS concentrations. Baseline estimates of estuarine TSS concentrations are presented in *Appendix E* in *Table E-1* through *Table E-4*. Concentrations varied from below 1 mg/L (one Southeastern and multiple Northeastern estuaries) to a maximum of 833 mg/L (San Antonio Bay, TX), and averaged 71.5 mg/L over all estuaries.

SPARROW-simulated TSS concentrations were compared to measured values from monitoring data. Measured observations were obtained from the EPA National Coastal Assessment (http://www.epa.gov/emap/nca/html/data/index.html), and augmented for San Francisco Bay with data from USGS (http://sfbay.wr.usgs.gov/access/wqdata/). Estimated TSS concentrations obtained from SPARROW predictions and application of the DCP approach in general exceed measured values, although agreement is remarkably good in many instances. In particular, DCP estimates of TSS for Delaware Bay (40.0 mg/L) are very close to observed values (38.45 mg/L), and virtually identical for Long Island Sound (9.7 mg/L predicted vs. 9.8 mg/L observed). Results for San Francisco Bay illustrate the difficulty in comparing model predictions with observations. DCP TSS estimates of 89.3 mg/L are slightly below the NOAA EMAP average of 71 observations (132 mg/L); but above the mean of the larger USGS sample (34 mg/L). Results suggest that SPARROW sediment loading predictions, transformed by the DCP approach, provide a reasonable basis for evaluating changes in TSS due to specific regulatory options, although predicted TSS values may differ from physical observation in absolute value in certain settings.

In the final analysis, the DCP approach was only used for the 43 reaches without flow predictions in SPARROW. The number of reaches in each estuary where the DCP approach was used to calculate TSS concentrations is shown in *Appendix E* in *Table E-5* through *Table E-8* .

6.4 Modeling Changes in Nutrient Concentrations

Reducing soil erosion and sediment runoff from construction sites not only reduces sediment loadings into streams, but can also reduce concentration of other pollutants that are present in the soil. To evaluate the water quality benefits of these ancillary reductions, EPA developed an approach that relies on the empirical relationship observed between baseline in-stream suspended sediment concentrations and TN and TP concentrations to derive estimates of the post-compliance nutrient reductions.

In evaluating potential approaches to estimating in-stream nutrient concentration reductions EPA faced several key challenges: (1) locating nationally available data on the nutrient content of soil or sediment at sufficient resolution; (2) accounting for the impact of the sediment delivery processes from edge of site to the stream on nutrient concentrations; (3) and estimating total nutrients as compared with particulate, dissolved, and other forms of nutrients. These challenges are in addition to the overall challenge of modeling the complexity of the nitrogen or phosphorus cycles and delivery processes, on a national scale. Based on literature, EPA determined that most of the total nitrogen and phosphorus in sediment associated with runoff are in the solid phase and therefore there is little difference between total and solid phase nutrient concentrations (Schuman et al. 1973a, b, as cited in Sharpley 1985; Haith 2009). It is estimated that up to approximately 90 percent of nutrients delivered to streams are bound to sediments (Schuman et al. 1973a, b, as cited in Sharpley 1985; Daniel et al. 1979); this is particularly true in the case of phosphorus and organic nitrogen. Several studies of construction site discharges have shown high correlations between sediment discharges and nutrients discharges (Daniel et al. 1979; Novotny and Chesters 1989; Harbor et al. 1995). Studies also indicate the ability of several construction site sediment erosion and control technologies to reduce nutrient (and other pollutants, e.g., metal, hydrocarbons, bacteria) discharges from construction sites (Horner et al. 1990; Stahre and Urbanos 1990; Bhaduri et al. 1995; Harbor et al. 1995; Lentz et al. 2002; USEPA 2006b; Bachand et al. 2007; Faucette et al. 2008; USEPA 2009b). Therefore, EPA assumed that the change in sediment-bound TN and TP would be representative of the overall change in TN and TP delivered to streams. While this simplification does not account for the complex processes that control nutrient fluxes through streams, EPA believes that it still provides reasonable estimates of the scale of changes that may be directly related to changes in sediment discharge from construction sites.

6.4.1 Modeling Approach Based on SPARROW Nutrient Models

USGS developed SPARROW models for TN and TP transport for the nontidal coterminous United States as described in Smith et al. (1997). As with the sediment model, the SPARROW nutrient models regression equations relate measured nutrient transport rates in streams to spatially referenced descriptors of pollution sources and land-surface and stream-channel characteristics. Observed TN and TP transport rates were derived from water-quality records for 414 stations in the National Stream Quality Accounting Network. Nutrient sources identified in the regression equations include point sources, applied fertilizer, livestock waste, nonagricultural land, and atmospheric deposition (TN model only). Surface characteristics found to be significant predictors of land-water delivery include soil permeability, stream density, and air temperature (TN model only).

The delivery processes, explanatory variables, and their coefficients are slightly different in each of the three SPARROW models (sediment, TN, and TP). As a result, it was not possible to utilize the nutrient models to predict changes in nutrient concentrations associated with construction sediment discharge by specifying the same input parameters used in the SPARROW sediment model. However, since the three models are based on the same hydrological network and baseline conditions, it is possible to relate predicted concentrations at the level of individual stream reaches. In this application, EPA used the baseline estimates for the three models to determine ratios between TN and sediment and between TP and sediment that were then used in conjunction with the predicted change in sediment concentrations to determine the change in nutrient concentrations, as represented by the following equation

$$\Delta[nutrient]_{scenario} = \Delta[TSS]_{scenario} \times \frac{[nutrient]_{baseline}}{[TSS]_{baseline}} \qquad \text{(Eq. 6-1)}$$

where Δ[nutrient] represents the change in nutrient concentration associated with the change in TSS concentration (Δ[TSS]). The variable [nutrient] represents the in-stream concentration of either TN or TP.

The nutrient-to-TSS ratio ([nutrient]/[TSS]) is calculated using SPARROW model outputs of TSS, TN and TP mass flux related to land-based runoff for the baseline conditions, as indicated in the equations below. The ratios were calculated for each individual reach as follows (where IL = incremental sediment mass flux per reach):

$$\frac{[TN]_{baseline}}{[TSS]_{baseline}} = \frac{\left(IL_{TN,total} - IL_{TN,population-related} - IL_{TN,atmospheric\ deposition}\right)}{IL_{TSS,total}} \qquad \text{(Eq. 6-2)}$$

$$\frac{[TP]_{baseline}}{[TSS]_{baseline}} = \frac{\left(IL_{TP,total} - IL_{TP,population-related}\right)}{IL_{TSS,total}} \qquad \text{(Eq. 6-3)}$$

Note that, since the regulation specifically targets reducing stormwater runoff and the associated sediment and nutrients and therefore does not address other potential sources of nutrients, EPA excluded from the calculations of the nutrient-to-sediment ratios nutrient loadings associated with population and atmospheric deposition sources.[6] To mitigate the effects of outlier TSS concentration on the estimated nutrient-to-sediment ratio, the top 5 percent of nutrient-to-TSS ratios were replaced by the 95th percentile

6 An example of population-related sources is effluent from publicly owned treatment works (POTWs). Atmospheric deposition is only a source in the nitrogen model.

value of the distribution. This is similar to the approach used to summarize SPARROW-predicted TSS concentrations at the national level which replaces values above the 95th percentile with the 95th percentile value. National statistics of the reach-specific nutrient-to-sediment ratios are summarized in *Table 6-2*.

Table 6-2: Nutrient-to-Sediment Ratios

	Average	Median	Min	Max
TN/TSS	0.006	0.005	0.000	0.028
TP/TSS	0.001	0.001	0.000	0.003

EPA averaged the reach-level ratios to the HUC8 watershed level to obtain a nutrient-to-sediment ratio that represents the average relationship between in-stream nutrients and suspended sediments within each basin. EPA later uses these watershed-average ratios to estimate reach-specific changes in suspended sediment concentrations for each scenario to yield estimated reach-level changes in nutrient concentration.

EPA compared the ratios described above to values found in the literature, databases, and fate and transport models *(Table 6-3)*. As shown in the table, the nutrient-to-sediment ratios calculated as described above based on SPARROW model outputs fall within the range of values published in the literature for both TN and TP. While differences are expected between soil and in-stream sediment concentrations, the values for both range widely in the United States and are shown in *Table 6-3* for reference.

Table 6-3: Literature Values of Nitrogen and Phosphorus in Soil and Sediment

Data Source	Data Type	Soil/Sediment	Nitrogen (%)	Phosphorus (%)
Ratios used in this analysis based on SPARROW[1]	**Modeled values based on field data**	**Sediment**	**0 to 2.77**	**0 to 0.32**
Haith et al. (1996)	Field data	Sediment	0.30	0.13
Sednet (Wilkinson et al. 2004)	Field data	Subsoils	0.1	0.025
Tisdale et al. (1985)	Field data	Natural, top 1 ft	0.03 to 0.4	-
Haith et al. (1996)	Field data	Surface 30 cm	<0.05 to ≥0.2	≤0.09 to 0.68
Havlin et al. (1999)	Modeled values based on field data	Surface soil	-	0.005 to 0.15
NRCS (2008)[2]	Field data	Soil (varied)	0 to 3.57	0.0001 to 0.58
SWAT (2005)	Modeled values based on field data	Surface soil	min 0.007[3]	0.0025[4]

Notes: Shaded rows indicate ranges that reflect geographically-specific data. For example, the Natural Resources Conservation Service (NRCS 2008) dataset is based on over 4,000 and 3,500 data points for TN and TP concentrations in soil, respectively, with a distribution of samples across the continental United States. The data provide useful references when evaluating the range of ratios obtained from SPARROW outputs but are insufficient for extrapolation to a reach-specific analysis.

[1] Presented ranges are for reach-level values excluding top 5 percent.

[2] Range of soil values seen in the continental United States (TN: 4,176 data points; TP: 3564 data points).

[3] Nitrate only.

[4] Mineral pool concentrations only.

6.5 Pollutant Load Modeling

The general procedures that EPA used to estimate sediment load reductions from construction sites are described in EPA's *Development Document for Final Effluent Guidelines and Standards for the Construction and Development Category* (USEPA 2009b). The results of this analysis are also detailed in that document. This analysis only considers stormwater sediment discharges to receiving waters and does not analyze dry-weather increases in surface water sediment and turbidity levels from dewatering discharges, wind deposition, construction activity taking place in surface waters, groundwater seepage, and vehicle and equipment washwaters.

Table 6-4 presents a national summary of sediment loadings calculated by EPA for the baseline and four regulatory options, as well as reductions under each regulatory option. Option 1 is expected to result in a reduction of 34 percent of construction sediment discharges nationally, while Option 2, Option 3, and Option 4 are expected to result in reductions of 70 percent, 87 percent, and 77 percent, respectively. Total reductions under Option 2 are 1.80 million tons of sediment, 2.25 million tons under Option 3, and 1.98 million tons under Option 4 compared to 870,000 tons under Option 1.

Table 6-4: Baseline Construction Sediment Loading Summary and Post-Compliance Reductions

Scenario	Sediment Loading (million tons)	Sediment Discharge Reduction (million tons)	Percent Reduction from Baseline
Baseline	2.58	–	–
Option 1	1.71	0.87	34%
Option 2	0.78	1.80	70%
Option 3	0.34	2.25	87%
Option 4	0.60	1.98	77%

Table 6-5 presents estimates of the amount of sediment entering reaches from construction related activity under each of these four options, by EPA region. The table shows the baseline amount of sediment entering reaches as well as post-compliance conditions and reductions for a regulatory option.

Table 6-5: Construction Sediment Loading Summary and Reductions by Option and EPA Region

EPA Region	Baseline	Option 1			Option 2			Option 3			Option 4		
	Loading (mil. tons)	Loading (mil. tons)	Reduction (mil. tons)	Percent Reduction	Loading (mil. tons)	Reduction (mil. tons)	Percent Reduction	Loading (mil. tons)	Reduction (mil. tons)	Percent Reduction	Loading (mil. tons)	Reduction (mil. tons)	Percent Reduction
1	0.02	0.01	0.01	34%	0.01	0.01	69%	0.00	0.02	85%	0.01	0.01	58%
2	0.04	0.02	0.01	34%	0.01	0.02	69%	0.00	0.03	86%	0.01	0.02	69%
3	0.15	0.10	0.05	34%	0.05	0.11	70%	0.02	0.13	87%	0.04	0.11	73%
4	0.90	0.60	0.30	34%	0.27	0.63	70%	0.12	0.78	87%	0.22	0.67	75%
5	0.28	0.19	0.10	34%	0.09	0.20	70%	0.04	0.25	87%	0.07	0.21	74%
6	0.85	0.56	0.29	34%	0.25	0.60	70%	0.11	0.74	87%	0.16	0.69	81%
7	0.23	0.15	0.08	34%	0.07	0.16	70%	0.03	0.20	87%	0.05	0.18	79%
8	0.04	0.03	0.01	34%	0.01	0.03	70%	0.01	0.04	87%	0.01	0.03	78%
9	0.04	0.02	0.01	34%	0.01	0.03	70%	0.00	0.03	87%	0.01	0.03	77%
10	0.03	0.02	0.01	34%	0.01	0.02	68%	0.00	0.02	85%	0.01	0.01	56%
Total	**2.58**	**1.71**	**0.87**	**34%**	**0.78**	**1.80**	**70%**	**0.33**	**2.25**	**87%**	**0.60**	**1.98**	**77%**

Note: Due to rounding, percentage reduction may not match the shown (rounded) reduction divided by the shown (rounded) baseline.

6.6 Water Quality Modeling Results

This section presents the results of water quality modeling using the SPARROW model for sediment and the above described estimation approach for nutrients. The SPARROW sediment model simulations generate estimates of water sediment content as Suspended Sediment Concentration (SSC), as consistent with water quality gauging records used to estimate SPARROW sediment models. By contrast, ambient sediment concentrations measured as Total Suspended Solids (TSS) are required in the calculation of the water quality index (details of the water quality index can be found in *Chapter 10*). Although SSC and TSS are interpreted in a similar manner as measures of the concentration of suspended solid-phase material in surface water bodies, Gray et al. (2000) have determined that differences in laboratory protocols result in systematic differences between SSC and TSS when measuring a given ambient concentration of suspended sediment. SSC measurements are systematically higher than TSS, particularly when the sand component of total sediments exceeds roughly 25 percent of total (dry) sediment mass. Therefore, for the analysis described in this document, SPARROW estimates of SSC have been divided by a factor of 1.3 to obtain the corresponding TSS values.

Construction activity discharges impact the majority of RF1 reaches in the coterminous United States. During the nine-year period from 1992 through 2001, approximately 71 percent of RF1 reach watersheds contained some level of construction. Construction discharges differ from other types of industrial discharges in this respect. Other types of industrial dischargers tend to be fewer in number and less widely distributed across the United States.

The intensity of construction activity, however, varies widely among individual RF1 watersheds. Approximately 43,900 RF1 watersheds had a net increase in urban acreage during the 1992-2001 time period. Approximately 17,200 RF1 watersheds had no change (~16,400 watersheds) or a minor decrease (~800 watersheds) in urban acreage. *Figure 6-4* illustrates the highly uneven distribution of construction acres among RF1 watersheds during the 1992-2001 time period. Watersheds represented in the figure account for 93 percent of all construction acres during the period, as shown in *Figure 6-5*, which shows the cumulative distribution of acres of construction by RF1 watersheds in decreasing order of construction acres by watershed and includes the national total number of acres of construction during the period. As the two figures show, relatively few watersheds account for a large fraction of acres of construction nationally. In *Figure 6-5,* for example, the 5 percent of RF1 watersheds containing the largest number of construction acres per watershed encompassed, as a group, over 2.4 million construction acres, or nearly 50 percent of all construction acreage for the 1992-2001 time period.

Figure 6-4: Plot of Construction Acres by RF1 Watershed Percentile Group (1992-2001)

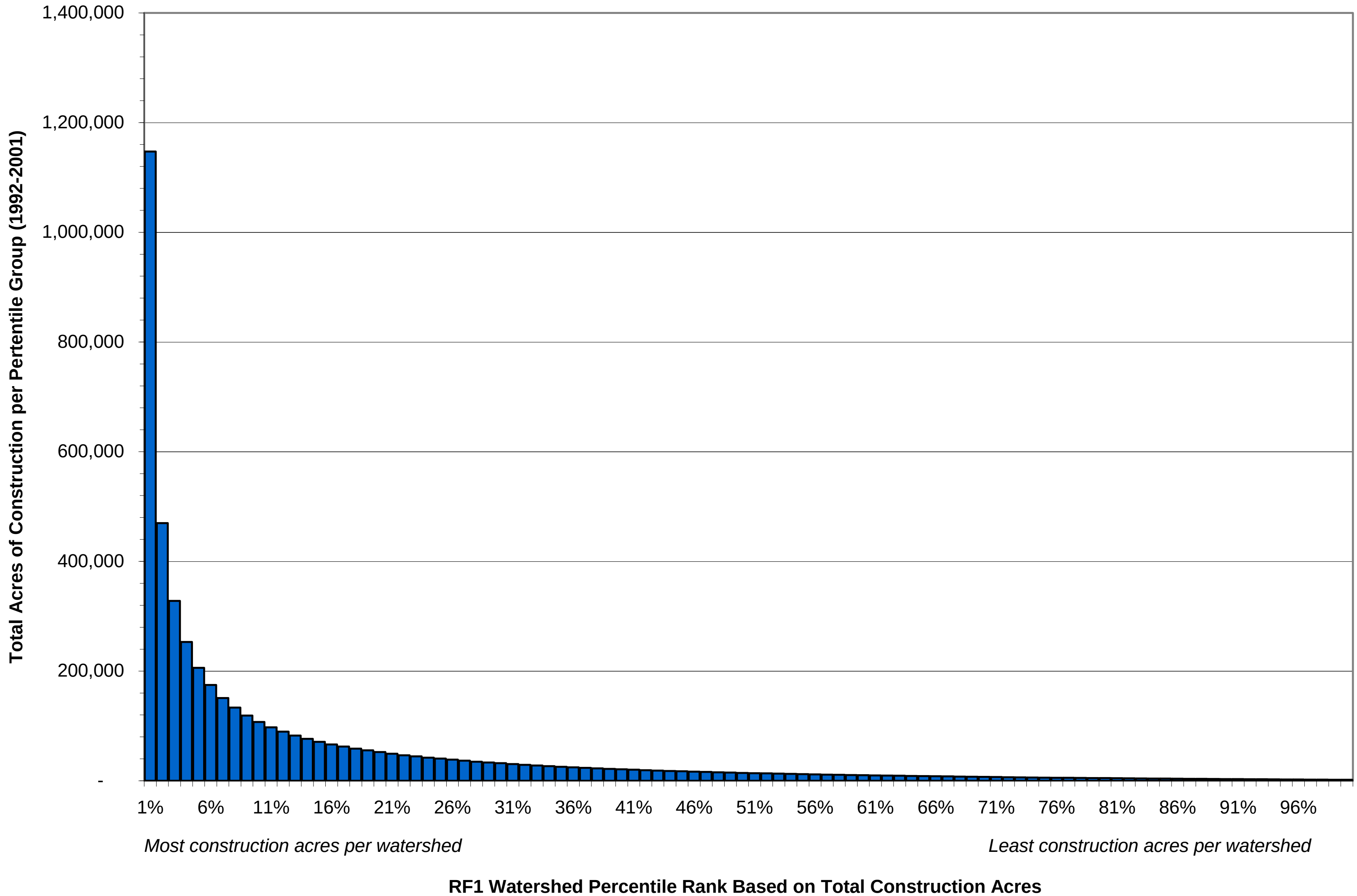


Figure 6-5: Cumulative Plot of Construction Acres by RF1 Watershed (1992-2001)

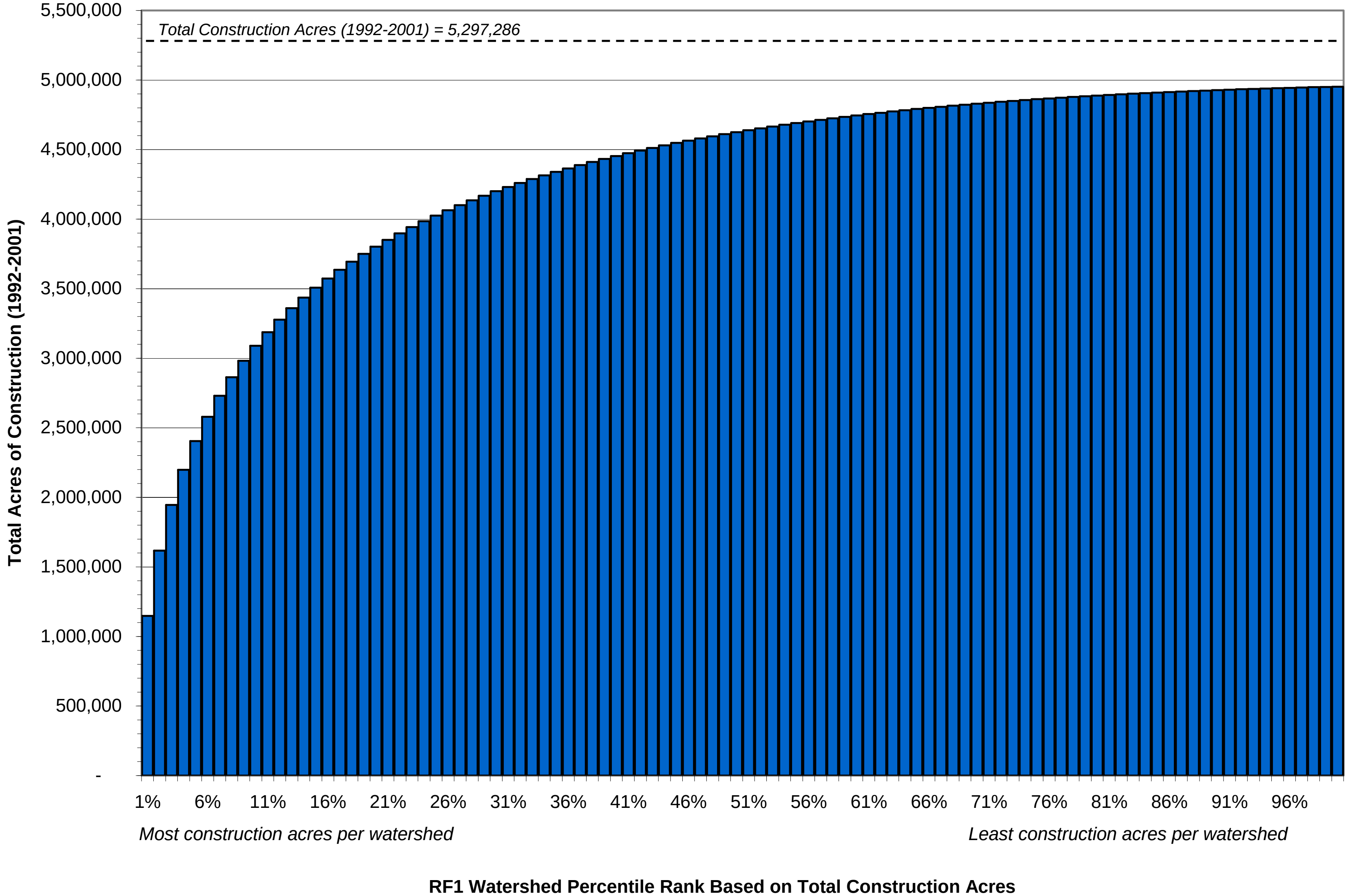


Figure 6-6 through *Figure 6-15* illustrate the distribution of construction acreage by EPA region.

Figure 6-6: EPA Region 1: Percent Urban Change 1992–2001 by RF1 Watershed

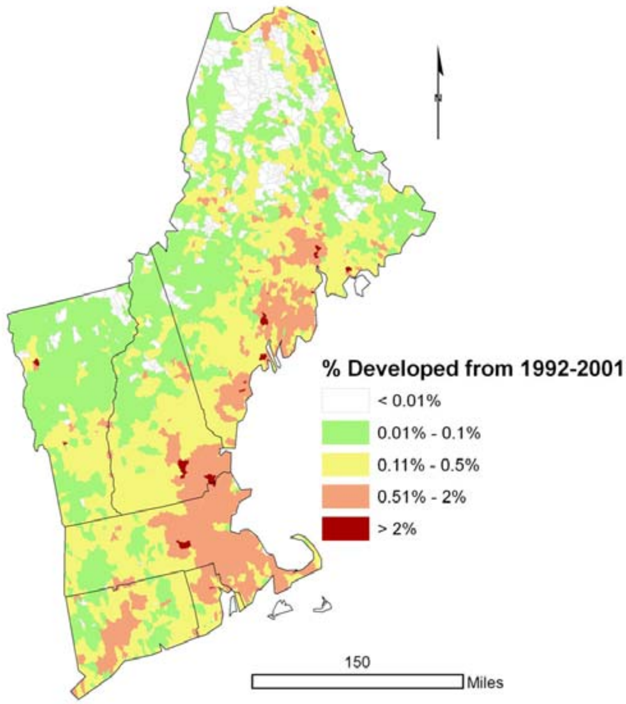


Figure 6-7: EPA Region 2: Percent Urban Change 1992–2001 by RF1 Watershed

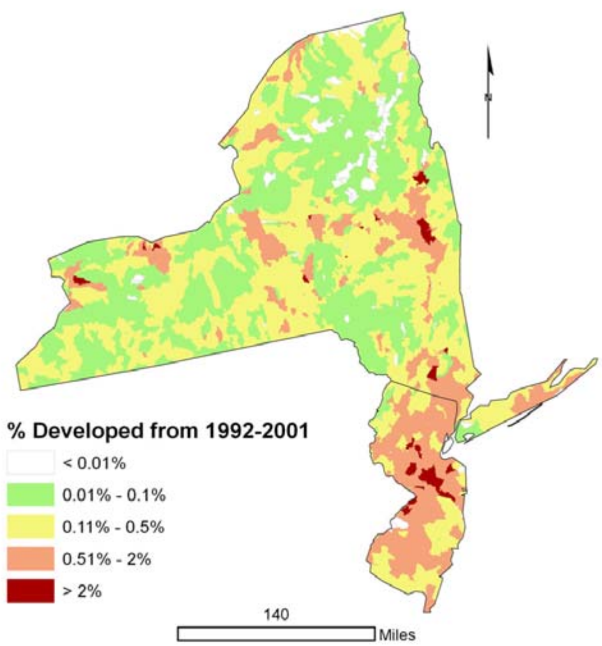


Figure 6-8: EPA Region 3: Percent Urban Change 1992–2001 by RF1 Watershed

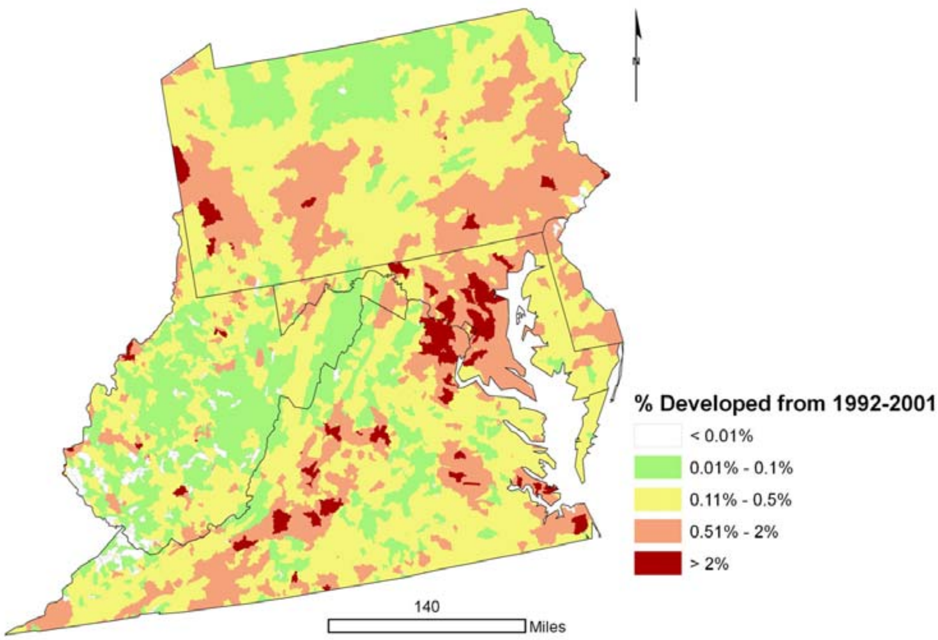


Figure 6-9: EPA Region 4: Percent Urban Change 1992–2001 by RF1 Watershed

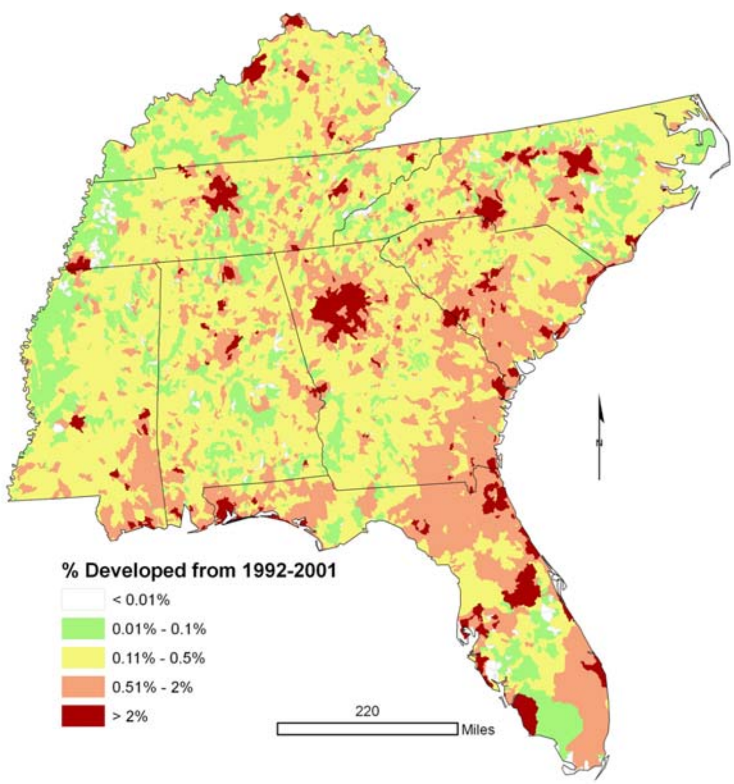


Figure 6-10: EPA Region 5: Percent Urban Change 1992–2001 by RF1 Watershed

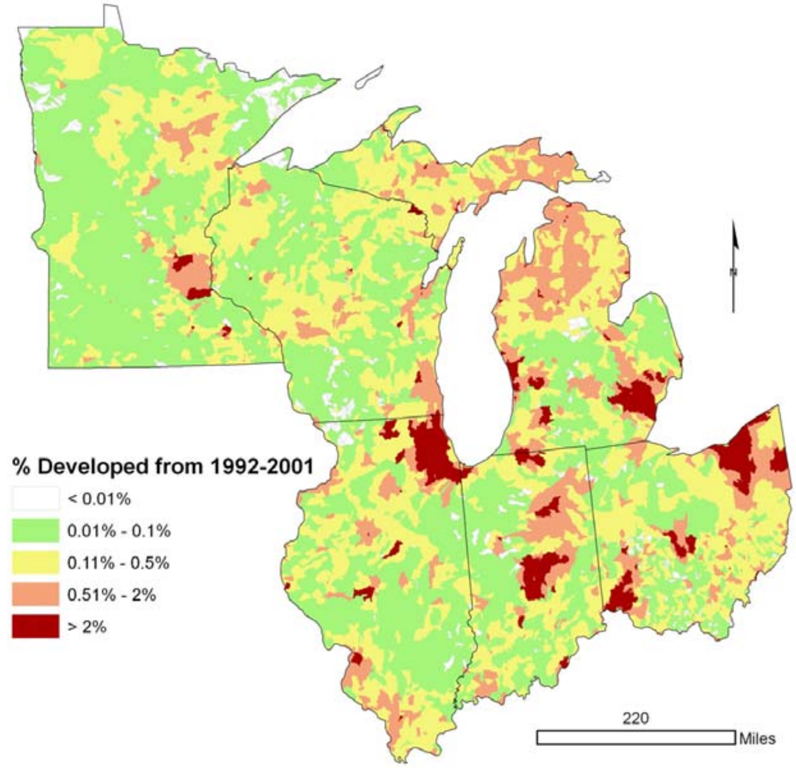


Figure 6-11: EPA Region 6: Percent Urban Change 1992–2001 by RF1 Watershed

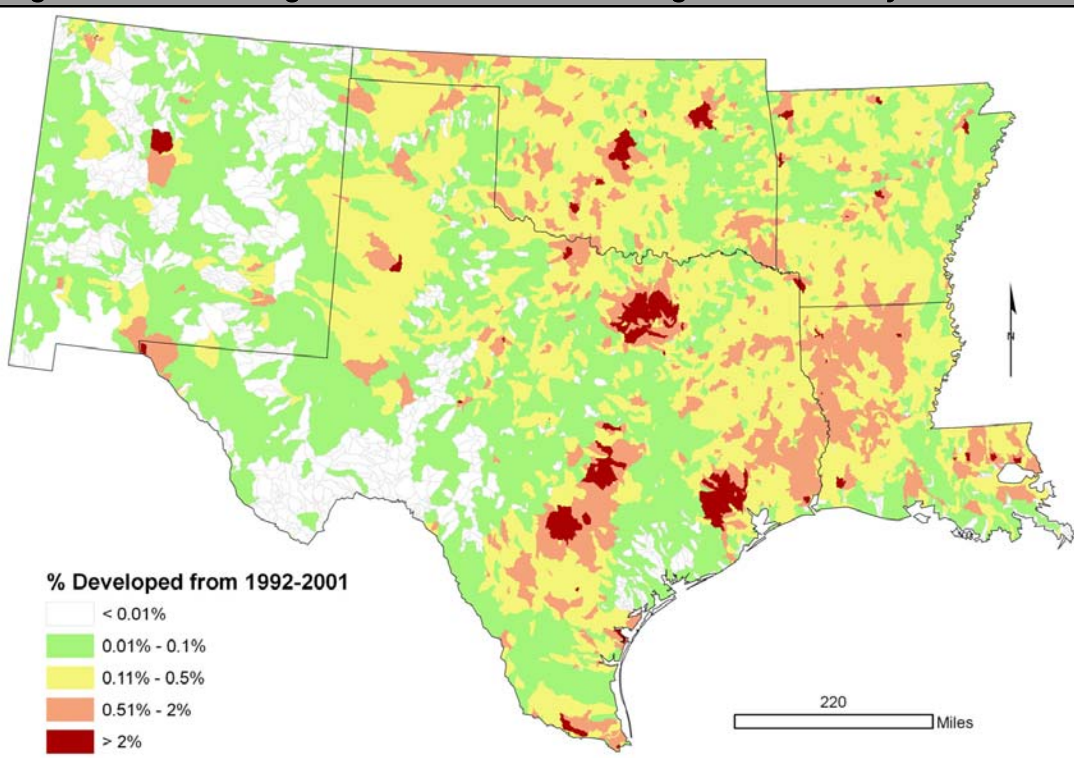


Figure 6-12: EPA Region 7: Percent Urban Change 1992–2001 by RF1 Watershed

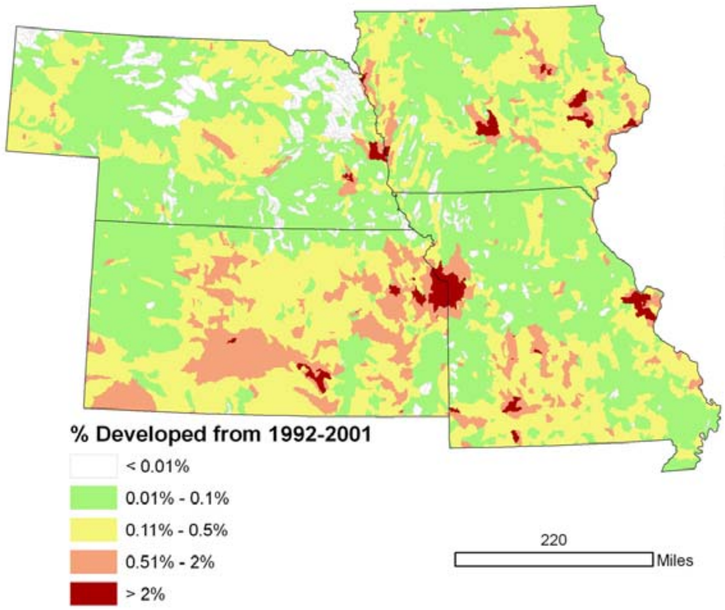


Figure 6-13: EPA Region 8: Percent Urban Change 1992–2001 by RF1 Watershed

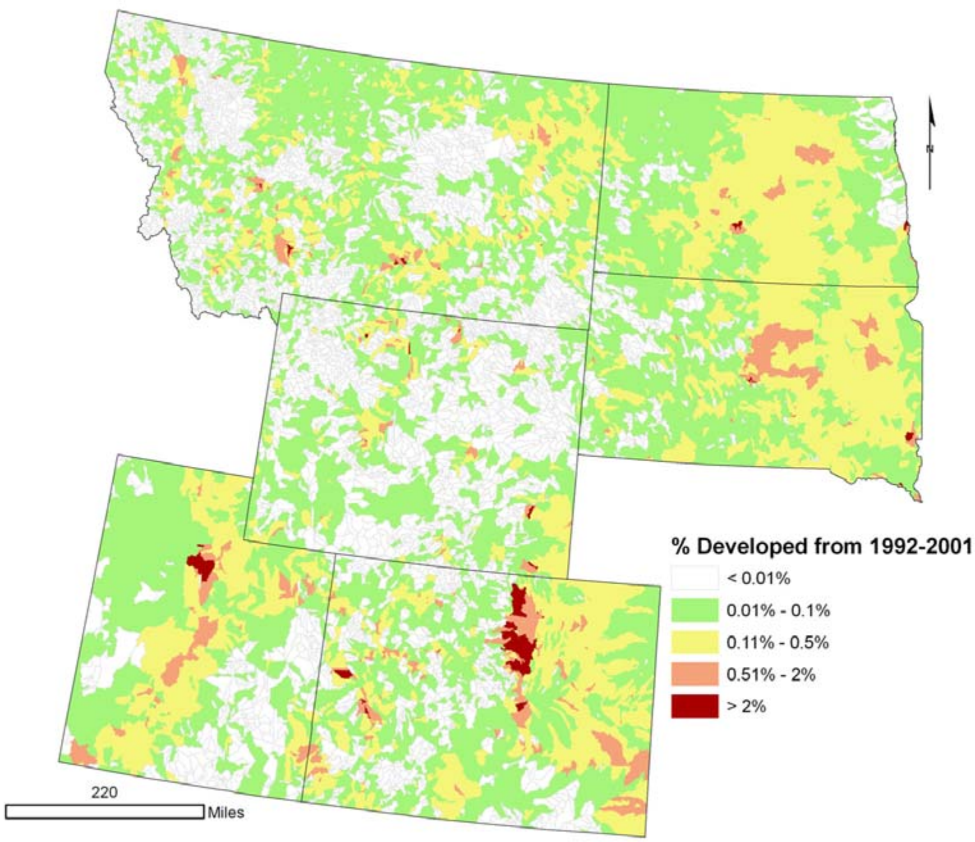


Figure 6-14: EPA Region 9: Percent Urban Change 1992–2001 by RF1 Watershed

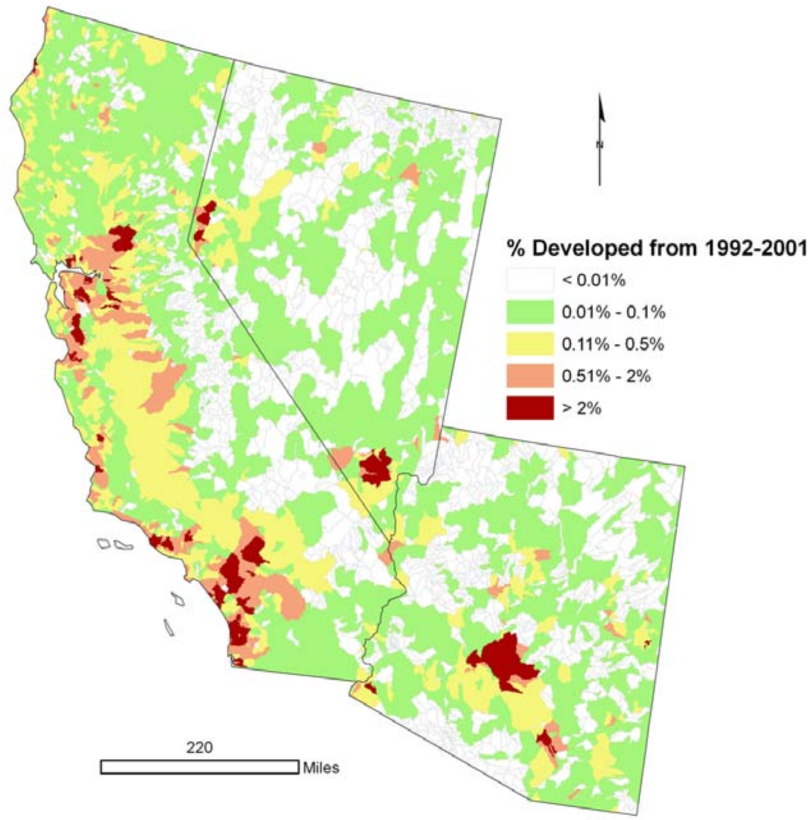
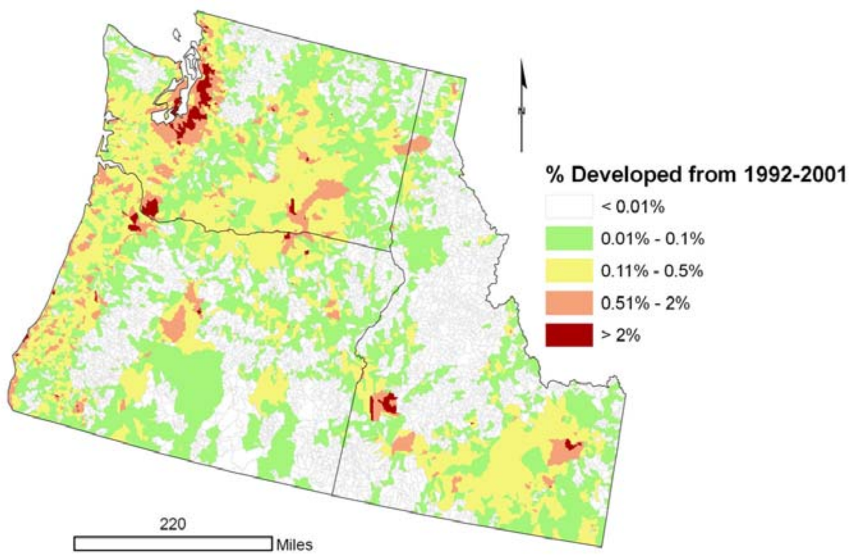


Figure 6-15: EPA Region 10: Percent Urban Change 1992–2001 by RF1 Watershed



Because construction sites are distributed so unevenly among watersheds, this chapter summarizes water quality information in two ways. One set of tables included in this chapter presents information for the relatively small number of RF1 watersheds that contain most construction acreage for the 1992-2001 time period. A second set of tables presents information for all RF1 reaches directly receiving discharges from construction sites, regardless of the level of construction activity in their watersheds.

An important factor to consider when examining the data summarized in the tables below is that episodic precipitation events are the primary cause of construction site discharges to surface waters. Most TSS and associated nutrient discharges from construction sites, therefore, take place during or shortly after precipitation events. Once a precipitation event ceases, discharges from construction sites generally cease within a short time period. The estimates presented in the tables below, however, are not intended to reflect surface water TSS and nutrient concentrations associated with individual, episodic storm events. They instead represent average concentrations estimated to exist in RF1 reaches over multi-year time periods. EPA uses these long-term estimates because the data and modeling resources currently available to EPA do not permit a finer level of time resolution.

EPA expects that, in general, surface water TSS and nutrient concentrations in the short time periods following storm events would be much higher than the concentrations presented in the tables below. TSS concentrations as high as 7,000 mg/L have been documented due to construction site discharges in downstream surface waters following precipitation events (Wolman and Schick 1967; Chisholm and Downs 1978; Downs and Appel 1986). Sediment and nutrient concentration reductions for the evaluated regulatory options would be higher, as well. During periods when there is no precipitation event to cause a discharge from a construction site, surface water sediment and nutrient concentrations would generally be lower.

A second important factor to consider when examining the data in the tables below is that construction site sediment and nutrient discharges also have strong geospatial variability. Geospatial differences occur as a result of variation in soil type, topography, distance from construction site to surface water, receiving surface water characteristics, and weather patterns, among other factors. These differences are reflected to a certain extent by the differences in surface water TSS and nutrient concentrations among different groups of reaches, as reflected by reaches having progressively greater construction acreage within their watershed (presented in *Table 6-16* through *Table 6-20*) and among the various EPA regions (presented in *Table 6-22* through *Table 6-31*). In general, however, the data presented below are averaged over large geographic regions because of limitations in the data and modeling resources currently available to EPA.

When summarizing TSS and nutrient concentrations, EPA calculated statistics based on reach-length weighted concentrations to account for the wide variation among reaches in length. EPA believes that this adjustment provides a more representative characterization of the distribution of concentrations by giving more weight to the longer reaches. Additionally, EPA has replaced concentrations that have concentrations above the 95th percentile (e.g., TSS greater than 6,157.8 mg/L under baseline conditions) with the 95th percentile value since these reaches could potentially be considered outliers. Finally, the data presented in the tables are for those reaches receiving *direct* discharges from construction sites. The decrease in loadings and concentrations in these reaches also translate into decreased concentrations in downstream reaches. While improvements in these downstream reaches are included in the benefits analysis, changes in concentration occurring downstream reaches that do not receive direct discharges from construction sites are not included in the tables in this chapter.

While summaries cannot adequately describe the full extent of temporal and geospatial variations in TSS and nutrient concentrations, they are nevertheless useful for interpreting broad-scale changes in water quality as a result of the evaluated regulatory options.

6.6.1 Current Water Quality Impacts to Surface Water from Construction Site Discharges

To estimate water quality impacts associated with construction site discharges under current conditions, EPA used SPARROW modeling of a hypothetical scenario in which all sediment discharges from construction activity are prevented from entering surface waters. Under this hypothetical scenario, loading from in-scope construction sites (one acre and greater in size) decrease by 2.58 million tons per year, relative to baseline conditions. It is important to note that a certain level of sediment is present in surface waters under natural, undisturbed conditions and that some sediment is necessary for the natural biological function of surface waters. Current levels of sediment discharge to the U.S. surface water network are much higher, however, than would be observed under natural, undisturbed conditions due to sediment discharges from construction, agriculture, and eroding streambeds associated with land use change and other human activity (see *Section 2.1)*. Therefore, while construction sediment discharges represent approximately 0.15 percent of total sediment in surface waters, they represent a greater percentage of total sediment pollution in the nation's waters. Given the relatively small percentage of U.S. land area dedicated to construction activity on an annual basis (~0.04 percent of coterminous United States), construction sites have a disproportionately high rate of sediment contribution to surface waters relative to other land uses.

Predicted changes, relative to baseline conditions, in TSS concentration for a hypothetical scenario with zero discharge from construction sites are presented in *Table 6-6* through *Table 6-9*. The tables show the distribution of TSS concentrations and reach miles improved. Average TSS concentrations decrease from 956.4 mg/L under baseline conditions to 954.1 mg/L under the zero discharge scenario. This represents a reduction of 2.4 mg/L or 0.3 percent. Median concentrations decrease from 289.3 to 287.5 mg/L, representing a change of 1.8 mg/L or 0.6 percent. In examining the spatial distribution of concentrations in the baseline and zero discharge scenarios, shown in *Table 6-7* and *Table 6-8*, respectively, the highest concentrations of TSS are seen in Regions 6, 7, 8, and 9, and this is also where the largest reductions are predicted. *Table 6-9* shows the distribution and magnitude of reductions in improved reach miles. Over 77 percent of improved reach miles are expected to have a decrease in TSS concentration that is greater than zero but less than 1 mg/L. Approximately 18 percent of improved reach miles will decrease in TSS concentration by 1 to 5 mg/L. Greater than 5 mg/L reductions are estimated in 4.2 percent of improved reach miles. These improvements are largely driven by Regions 4, 6, 7 and 10. Eliminating sediment discharge from construction sites also reduces sediment accumulation in reservoirs by 3.7 million pounds per year.

Table 6-6: Distribution of TSS Concentrations based on SPARROW Output for 31,927 RF1 Reaches Receiving Construction Sediment Discharges

Scenario	Reach Count	Reach Miles	Average TSS (mg/L)[1,2]	Reduction in Average TSS (mg/L)	Distribution of TSS Concentrations by Percentile[1,2] (mg/L)				
					5th	25th	50th	75th	95th
Baseline	31,927	412,062	956.4	–	29.7	109.9	289.3	880.0	6,157.8
Hypothetical No Discharge Scenario	31,927	412,062	954.1	2.4	28.6	108.7	287.5	877.9	6,156.8

[1]Reach-length weighted.
[2]Any TSS concentration above the 95th percentile is assumed to be an outlier and was replaced with the 95th percentile value.

Table 6-7: Summary of Baseline TSS Concentration in RF1 Reaches Receiving Construction Sediment Discharges, by EPA Region

EPA Region	Reach Count	Reach Miles	Average TSS (mg/L)[1,2]	Distribution of TSS Concentrations by Percentile[1,2] (mg/L)				
				5th	25th	50th	75th	95th
1	1,213	10,740	77.5	9.8	25.2	41.2	57.2	118.7
2	1,111	12,560	110.2	10.5	38.8	79.0	135.7	283.1
3	2,012	25,471	180.6	44.5	83.6	141.7	228.3	447.4
4	7,564	83,058	236.8	33.2	91.4	180.6	311.1	596.8
5	3,918	56,909	296.0	20.6	60.9	196.1	430.4	880.4
6	4,634	70,687	1,567.7	82.9	259.5	641.6	2,137.4	6,157.8
7	3,063	46,903	1,800.8	177.6	484.7	1,050.3	2,008.6	6,157.8
8	4,300	57,513	2,024.6	79.2	359.7	1,131.2	3,104.9	6,157.8
9	1,439	22,280	1,486.7	31.6	157.6	434.1	1,932.3	6,157.8
10	2,673	25,940	239.6	20.6	52.6	108.2	233.1	882.4
Nation	**31,927**	**412,062**	**956.4**	**29.7**	**109.9**	**289.3**	**880.0**	**6,157.8**

[1]Reach-length weighted.
[2]Any TSS concentration above the 95th percentile is assumed to be an outlier and was replaced with the 95th percentile value.

Table 6-8: Summary of TSS Concentrations Under the Hypothetical No Discharge Scenario, by EPA Region

EPA Region	Reach Count	Reach Miles	Average TSS (mg/L)[1,2]	Distribution of TSS Concentrations by Percentile[1,2] (mg/L)				
				5th	25th	50th	75th	95th
1	1,213	10,740	77.3	9.7	25.2	41.1	57.1	118.6
2	1,111	12,560	110.2	10.5	38.8	79.0	135.5	281.7
3	2,012	25,471	180.4	44.4	83.6	141.5	228.2	447.3
4	7,564	83,058	235.3	32.2	90.4	178.8	310.1	590.9
5	3,918	56,909	295.8	20.5	60.9	195.8	430.4	880.3
6	4,634	70,687	1,562.0	75.1	254.0	634.6	2,123.9	6,156.8
7	3,063	46,903	1,797.9	173.9	481.7	1,048.3	2,007.3	6,156.8
8	4,300	57,513	2,017.5	75.4	354.5	1,126.6	3,104.2	6,156.8
9	1,439	22,280	1,480.9	29.8	152.6	430.2	1,900.9	6,156.8
10	2,673	25,940	238.4	19.4	51.4	107.5	233.1	882.4
Nation	**31,927**	**412,062**	**954.1**	**28.6**	**108.7**	**287.5**	**877.9**	**6,156.8**

[1]Reach-length weighted.
[2]Any TSS concentration above the 95th percentile is assumed to be an outlier and was replaced with the 95th percentile value.

Table 6-9: Water Quality Impacts: Improvements in TSS Concentrations Under the Hypothetical No Discharge Scenario, by EPA Region

EPA Region	Total Improved Reach Miles[1]	Range of Reductions in TSS Concentration (mg/L)									
		0 < Δ TSS < 1		1 ≤ Δ TSS < 5		5 ≤ Δ TSS < 10		10 ≤ Δ TSS < 50		50 ≤ Δ TSS	
		Reach Miles	% of Total	Reach Miles	% of Total	Reach Miles	% of Total	Reach Miles	% of Total	Reach Miles	% of Total
1	10,740	10,421	97.0%	219	2.0%	64	0.6%	36	0.3%	0	0.0%
2	12,560	12,488	99.4%	72	0.6%	0	0.0%	0	0.0%	0	0.0%
3	25,471	24,256	95.2%	1,073	4.2%	112	0.4%	30	0.1%	0	0.0%
4	83,028	64,885	78.1%	14,320	17.2%	2,236	2.7%	1,401	1.7%	186	0.2%
5	56,906	54,801	96.3%	1,974	3.5%	97	0.2%	34	0.1%	0	0.0%
6	70,630	33,122	46.9%	27,840	39.4%	4,506	6.4%	4,501	6.4%	661	0.9%
7	46,859	31,518	67.3%	13,896	29.7%	816	1.7%	584	1.2%	44	0.1%
8	57,322	47,070	82.1%	9,967	17.4%	227	0.4%	49	0.1%	10	0.0%
9	22,204	17,373	78.2%	4,319	19.4%	283	1.3%	200	0.9%	29	0.1%
10	25,920	23,283	89.8%	1,527	5.9%	559	2.2%	478	1.8%	72	0.3%
Nation	**411,640**	**319,218**	**77.5%**	**75,206**	**18.3%**	**8,900**	**2.2%**	**7,314**	**1.8%**	**1,002**	**0.2%**

[1] Does not include 40 freshwater reaches (422 reach miles) for which SPARROW does not predict a concentration.

Table 6-10 through *Table 6-15* present the reductions in TN and TP concentrations associated with reduced sediment discharge under the hypothetical zero discharge scenario. As described in *Section 6.4*, reductions in TN and TP concentrations are calculated based on the predicted reduction of in-stream sediment concentrations. The distribution of improved reach miles for TN and TP is therefore the same as for TSS, which is presented above in *Table 6-9*.

Reductions in average TN are predicted to be 0.02 mg/L or 0.1 percent. Reductions in the median concentrations are 0.01 mg/L or 0.1 percent and 95th percentile concentrations with 0.04 mg/L or 0.3 percent reductions.

Reductions in average TP concentrations are predicted to be 6.0 µg/L or 0.2 percent. Reductions in the median concentrations are 2.2 µg/L or 0.1 percent and 95th percentile concentrations with 22.3 µg/L or 0.6 percent reductions.

The geographic distribution of TN and TP concentrations is similar to TSS concentrations, with the highest concentrations seen in Regions 6, 7, 8, and 9. In general, these regions are also predicted to see the largest reductions in concentrations.

Table 6-10: Distribution of TN Concentrations based on SPARROW Output for 31,927 RF1 Reaches Receiving Construction Sediment Discharges

Scenario	Reach Count	Reach Miles	Average TN (mg/L)[1,2]	Reduction in Average TN (mg/L)	Distribution of TN Concentrations by Percentile (mg/L) [1,2]				
					5th	25th	50th	75th	95th
Baseline	31,927	412,062	18.22	–	0.37	0.91	1.65	3.88	14.95
Hypothetical No Discharge Scenario	31,927	412,062	18.21	0.02	0.36	0.90	1.64	3.86	14.91

[1]Reach-length weighted.
[2]Nutrient reductions are based on TSS reductions. Any TSS concentration above the 95th percentile is assumed to be an outlier and was replaced with the 95th percentile value.

Table 6-11: Summary of Baseline TN Concentration in RF1 Reaches Receiving Construction Sediment Discharges, by EPA Region

EPA Region	Reach Count	Reach Miles	Average TN (mg/L)[1]	Distribution of TN Concentrations by Percentile (mg/L) [1]				
				5th	25th	50th	75th	95th
1	1,213	10,740	1.32	0.24	0.59	0.85	1.27	2.57
2	1,111	12,560	19.43	0.33	0.95	1.37	1.98	3.85
3	2,012	25,471	1.88	0.61	1.02	1.35	1.95	4.35
4	7,564	83,058	4.39	0.29	0.70	1.06	1.59	3.06
5	3,918	56,909	7.76	0.51	1.49	3.68	7.50	15.86
6	4,634	70,687	26.15	0.45	1.07	1.83	3.26	12.38
7	3,063	46,903	22.87	1.10	2.64	5.11	9.56	21.50
8	4,300	57,513	40.01	0.50	1.08	2.44	5.99	28.66
9	1,439	22,280	46.56	0.24	0.55	1.28	3.35	54.43
10	2,673	25,940	5.30	0.23	0.49	0.83	1.58	5.29
Nation	**31,927**	**412,062**	**18.22**	**0.37**	**0.91**	**1.65**	**3.88**	**14.95**

[1]Reach-length weighted.

Table 6-12: Summary of TN Concentrations Under the Hypothetical No Discharge Scenario, by EPA Region

EPA Region	Reach Count	Reach Miles	Average TN (mg/L)[1,2]	Distribution of TN Concentrations by Percentile (mg/L) [1,2]				
				5th	25th	50th	75th	95th
1	1,213	10,740	1.32	0.23	0.59	0.85	1.27	2.57
2	1,111	12,560	19.43	0.33	0.95	1.37	1.98	3.85
3	2,012	25,471	1.88	0.61	1.02	1.35	1.95	4.34
4	7,564	83,058	4.38	0.28	0.70	1.06	1.58	3.06
5	3,918	56,909	7.75	0.51	1.48	3.68	7.49	15.86
6	4,634	70,687	26.11	0.42	1.06	1.82	3.23	12.33
7	3,063	46,903	22.85	1.08	2.61	5.10	9.49	21.50
8	4,300	57,513	39.99	0.48	1.07	2.42	5.96	28.64
9	1,439	22,280	46.51	0.23	0.54	1.28	3.26	54.43
10	2,673	25,940	5.29	0.23	0.49	0.83	1.58	5.29
Nation	**31,927**	**412,062**	**18.21**	**0.36**	**0.90**	**1.64**	**3.86**	**14.91**

[1]Reach-length weighted.
[2] Nutrient reductions are based on TSS reductions. Any TSS concentration above the 95th percentile is assumed to be an outlier and was replaced with the 95th percentile value.

Table 6-13: Distribution of TP Concentrations based on SPARROW Output for 31,927 RF1 Reaches Receiving Construction Sediment Discharges

Scenario	Reach Count	Reach Miles	Average TP (µg/L)[1,2]	Reduction in Average TP (µg/L)	Distribution of TP Concentrations by Percentile (µg/L) [1,2] 5th	25th	50th	75th	95th
Baseline	31,927	412,062	3,921.7	–	30.5	104.4	253.2	685.3	2,907.3
Hypothetical No Discharge Scenario	31,927	412,062	3,915.6	6.0	29.5	103.3	251.0	683.2	2,885.0

[1]Reach-length weighted.
[2] Nutrient reductions are based on TSS reductions. Any TSS concentration above the 95th percentile is assumed to be an outlier and was replaced with the 95th percentile value.

Table 6-14: Summary of Baseline TP Concentration in RF1 Reaches Receiving Construction Sediment Discharges, by EPA Region

EPA Region	Reach Count	Reach Miles	Average TP (µg/L)[1]	Distribution of TP Concentrations by Percentile (µg/L) [1] 5th	25th	50th	75th	95th
1	1,213	10,740	109.5	9.9	29.6	49.2	86.5	233.8
2	1,111	12,560	1,697.5	16.6	47.2	91.2	152.1	403.0
3	2,012	25,471	209.3	37.7	80.2	135.4	233.5	633.6
4	7,564	83,058	885.6	30.0	81.8	139.5	241.9	547.4
5	3,918	56,909	651.3	18.6	75.6	225.3	475.9	1,246.4
6	4,634	70,687	8,110.6	63.4	208.4	519.5	1,216.9	5,557.8
7	3,063	46,903	3,380.2	140.6	377.8	603.9	1,012.7	2,853.1
8	4,300	57,513	7,282.3	69.0	249.5	659.0	1,449.5	5,803.0
9	1,439	22,280	13,544.9	37.0	114.9	489.0	1,652.3	30,733.9
10	2,673	25,940	965.7	22.1	56.3	113.4	303.0	1,318.7
Nation	**31,927**	**412,062**	**3,921.7**	**30.5**	**104.4**	**253.2**	**685.3**	**2,907.3**

[1]Reach-length weighted.

Table 6-15: Summary of TP Concentrations Under the Hypothetical No Discharge Scenario, by EPA Region

EPA Region	Reach Count	Reach Miles	Average TP (µg/L)[1,2]	Distribution of TP Concentrations by Percentile (µg/L) [1,2] 5th	25th	50th	75th	95th
1	1,213	10,740	109.3	9.9	29.4	49.1	86.5	233.7
2	1,111	12,560	1,697.5	16.6	47.2	91.1	151.9	402.9
3	2,012	25,471	209.1	37.7	80.0	135.1	233.2	633.4
4	7,564	83,058	884.4	29.2	81.1	138.5	239.9	547.3
5	3,918	56,909	651.1	18.5	75.5	225.3	475.8	1,246.4
6	4,634	70,687	8,094.3	57.6	202.5	516.2	1,207.3	5,557.7
7	3,063	46,903	3,376.8	138.4	374.8	601.9	1,008.8	2,850.7
8	4,300	57,513	7,277.0	66.6	246.4	656.6	1,445.2	5,780.1
9	1,439	22,280	13,512.6	35.3	111.2	474.1	1,644.4	30,733.9
10	2,673	25,940	964.6	21.0	55.4	112.2	297.9	1,267.0
National	**31,927**	**412,062**	**3,915.6**	**29.5**	**103.3**	**251.0**	**683.2**	**2,885.0**

[1]Reach-length weighted.
[2] Nutrient reductions are based on TSS reductions. Any TSS concentration above the 95th percentile is assumed to be an outlier and was replaced with the 95th percentile value.

6.6.2 Estimated Changes in TSS Concentrations in RF1 Watersheds with Most Developed Acres (1992-2001) Receiving Sediment Loading from Construction Sites

Table 6-16 summarizes the characteristics of different groups of modeled watersheds in terms of the relative amount of development they contain. The table provides information for three particular subgroups that represent the "top" watersheds in terms of the amount of developed land during the period of 1992-2001: top 1 percent, top 10 percent, and top 25 percent. As discussed above, development is not evenly distributed during the nine-year period under examination and each of the watershed subsets therefore account for a relatively disproportionate share of construction acres. For example, the top 10 percent of watersheds account for over 58 percent of the increase in developed land in the coterminous United States, while the top 25 percent account for 76 percent of all developed land. These watersheds are also distributed unevenly across EPA regions both in terms of total reach miles and the number of waterbody reaches, as shown in *Table 6-17*. For example, approximately 22 percent of reach miles within the top 1 percent are found in Region 5 even though the region accounts for approximately 14 percent of overall RF1 reach miles.

These differences are reflected in TSS concentration estimates shown in *Table 6-16* through *Table 6-20* for each of the three top groups, respectively, and across the four regulatory options (see *Chapter 1* for option descriptions). As shown in the tables, the top 1 percent of watersheds in terms of construction acres tend to have the highest sediment concentrations, both under current conditions and under the four options. While the three top groups result in a similar fraction of miles showing improvements in TSS concentrations across the three groups, reaches within the top 1 percent of watersheds exhibit a higher *magnitude* of TSS concentration reductions than reaches in watersheds that contain comparatively fewer construction acres.

Table 6-16: Reaches with Largest Increase in Developed Land Area (1992-2001)

Percent of RF1 Watersheds[1]	Number of RF1 Watersheds[2]	% of Total Increase in Coterminous U.S. Developed Land	Total Increase in Developed Land Area (1992-2001) (million acres)	Average Increase in Developed Land Area per RF1 Watersheds (1992-2001) (acres)	Total Land Area (million acres)	% of Coterminous U.S. Land Area
Top 1% of RF1 Watersheds	319	21.7%	1.1	3,596.4	52.6	2.8%
Top 10% of RF1 Watersheds	3,192	58.3%	3.1	968.0	305.2	16.1%
Top 25% of RF1 Watersheds	7,981	76.0%	4.0	504.4	572.9	30.3%
All RF1 Watersheds in Coterminous U.S. (100% of Reaches)	31,927	93.5%	5.0	155.1	1,291.9	68.2%

[1] Top 1% of RF1 watersheds had 1,841 acres or greater of developed land area (1992-2001); Top 10% of watersheds had 321 acres or greater of developed land area (1992-2001); Top 25% of watersheds had 124 acres or greater of developed land area (1992-2001).

[2] RF1 watersheds included in count are those that receive construction loadings *and* for which both NLCD and SPARROW data are available.

Table 6-17: Distribution of RF1 Watersheds that Receive Direct Loadings by Increase in Developed Land Area (1992-2001), by EPA Region

EPA Region	Top 1% of Watersheds		Top 10% of Watersheds		Top 25% of Watersheds		Coterminous U.S. (100% of Reaches)	
	Reach Miles	% of Miles	Reach Miles	% of Miles	Reach Miles	% of Miles	Reach Miles	% of Miles
1	21	0%	911	1%	2,088	1%	10,740	3%
2	50	1%	1,502	2%	3,646	3%	12,560	3%
3	410	6%	4,582	7%	9,774	7%	25,471	6%
4	1,472	20%	14,276	22%	33,331	24%	83,058	20%
5	1,590	22%	9,733	15%	19,670	14%	56,909	14%
6	1,540	21%	15,428	24%	31,300	22%	70,687	17%
7	736	10%	7,042	11%	16,350	12%	46,903	11%
8	450	6%	4,758	7%	13,362	9%	57,513	14%
9	947	13%	3,915	6%	6,827	5%	22,280	5%
10	172	2%	2,139	3%	5,239	4%	25,940	6%
Nation	**7,387**	**100%**	**64,284**	**100%**	**141,587**	**100%**	**412,062**	**100%**

Table 6-18: Estimated Changes in RF1 Reach TSS Concentration by Policy Option: Top 1% of RF1 Watersheds Receiving Direct Construction Loadings by Increase in Developed Land Area (1992-2001)

Scenario	Median TSS (mg/L)[1]	Average TSS (mg/L)[1]	Total Improved RF1 Miles	Total Improved % of Total Miles[2]	$0 < \Delta$ TSS < 1 RF1 Miles	$0 < \Delta$ TSS < 1 % of Total Improved Miles	$1 \leq \Delta$ TSS < 5 RF1 Miles	$1 \leq \Delta$ TSS < 5 % of Total Improved Miles	$5 \leq \Delta$ TSS < 10 RF1 Miles	$5 \leq \Delta$ TSS < 10 % of Total Improved Miles	$10 \leq \Delta$ TSS < 50 RF1 Miles	$10 \leq \Delta$ TSS < 50 % of Total Improved Miles	$50 \leq \Delta$ TSS RF1 Miles	$50 \leq \Delta$ TSS % of Total Improved Miles
Baseline	425.6	1,073.3	–	–	–	–	–	–	–	–	–	–	–	–
Option 1	425.2	1,069.2	6,932	93.8%	4,891	70.6%	1,288	18.6%	354	5.1%	381	5.5%	19	0.3%
Option 2	424.9	1,067.0	6,932	93.8%	4,242	61.2%	1,509	21.8%	416	6.0%	683	9.9%	82	1.2%
Option 3	424.7	1,065.9	7,372	99.8%	4,457	60.5%	1,418	19.2%	618	8.4%	748	10.1%	131	1.8%
Option 4	424.8	1,066.5	6,932	93.8%	4,184	60.4%	1,359	19.6%	511	7.4%	776	11.2%	103	1.5%

Note: Range of Reductions in TSS Concentrations (mg/L) spans the Δ TSS columns.

[1] River-mile weighted.

[2] Total miles (7,387) are based on top 1% of reaches by area of urban development from 1992 to 2001.

Table 6-19: Estimated Changes in RF1 Reach TSS Concentration by Policy Option: Top 10% of RF1 Watersheds Receiving Direct Construction Loadings by Increase in Developed Land Area (1992-2001)

Scenario	Median TSS (mg/L)[1]	Average TSS (mg/L)[1]	Total Improved RF1 Miles	Total Improved % of Total Miles[2]	$0 < \Delta$ TSS < 1 RF1 Miles	$0 < \Delta$ TSS < 1 % of Total Improved Miles	$1 \leq \Delta$ TSS < 5 RF1 Miles	$1 \leq \Delta$ TSS < 5 % of Total Improved Miles	$5 \leq \Delta$ TSS < 10 RF1 Miles	$5 \leq \Delta$ TSS < 10 % of Total Improved Miles	$10 \leq \Delta$ TSS < 50 RF1 Miles	$10 \leq \Delta$ TSS < 50 % of Total Improved Miles	$50 \leq \Delta$ TSS RF1 Miles	$50 \leq \Delta$ TSS % of Total Improved Miles
Baseline	289.1	947.7	–	–	–	–	–	–	–	–	–	–	–	–
Option 1	287.5	945.0	60,992	94.9%	48,452	79.4%	9,547	15.7%	1,708	2.8%	1,067	1.8%	218	0.4%
Option 2	287.0	943.6	60,992	94.9%	41,568	68.2%	13,413	22.0%	2,970	4.9%	2,674	4.4%	367	0.6%
Option 3	286.0	943.9	64,202	99.9%	42,342	66.0%	14,308	22.3%	3,643	5.7%	3,419	5.3%	490	0.8%
Option 4	286.8	943.3	60,992	94.9%	40,599	66. 6%	13,551	22.2%	3,355	5.5%	3,044	5.0%	443	0.7%

Note: Range of Reductions in TSS Concentrations (mg/L) spans the Δ TSS columns.

[1] River-mile weighted.

[2] Total miles (64,284) are based on top 10% of reaches by area of urban development from 1992 to 2001.

Table 6-20: Estimated Changes in RF1 Reach TSS Concentration by Policy Option: Top 25% of RF1 Watersheds by Increase in Developed Land Area (1992-2001)

					Range of Reductions in TSS Concentrations (mg/L)									
			Total Improved		0 < Δ TSS < 1		1 ≤ Δ TSS < 5		5 ≤ Δ TSS < 10		10 ≤ Δ TSS < 50		50 ≤ Δ TSS	
Scenario	Median TSS (mg/L)[1]	Average TSS (mg/L)[1]	RF1 Miles	% of Total Miles[2]	RF1 Miles	% of Total Improved Miles	RF1 Miles	% of Total Improved Miles	RF1 Miles	% of Total Improved Miles	RF1 Miles	% of Total Improved Miles	RF1 Miles	% of Total Improved Miles
Baseline	283.5	929.9	–	–	–	–	–	–	–	–	–	–	–	–
Option 1	282.6	928.2	134,556	95.0%	114,258	84.9%	16,259	12.1%	2,336	1.7%	1,406	1.1%	297	0.2%
Option 2	281.9	927.3	134,556	95.0%	100,071	74.4%	25,718	19.1%	4,638	3.5%	3,635	2.7%	493	0.4%
Option 3	281.5	926.8	141,498	99.9%	101,252	71.6%	29,060	20.5%	5,898	4.2%	4,630	3.3%	659	0.5%
Option 4	281.5	927.1	134,556	95.0%	97,345	72.3%	27,184	20.2%	5,288	3.9%	4,132	3.1%	608	0.5%

[1] River-mile weighted.

[2] Total miles (141,489) are based on top 25% of reaches by area of urban development from 1992 to 2001.

6.6.3 Estimated Changes in TSS, TN, and TP Concentrations for all RF1 Watersheds Receiving Loading from Construction Sites

Table 6-21 summarizes national average reach-length weighted TSS concentrations in RF1 reaches as estimated by SPARROW under both the baseline scenario representing current conditions and the four regulatory options (see *Chapter 1* for option descriptions). Information is presented for all reaches in the RF1 network that receive sediment loading from construction sites.

Table 6-21 also presents information on the distribution of TSS concentrations under the baseline scenario representing current conditions and the four regulatory options. TSS concentrations in the RF1 network vary over a wide range. Values for the 5th and 95th percentile bounds under baseline conditions are 29.7 and 6,157.8 mg/L respectively. Median (50th percentile) TSS concentrations are approximately 289.3 mg/L under the baseline scenario. Under Option 1, the median concentration decreases 0.8 mg/L to 288.6 mg/L. Median concentration under Option 2 decreases 1.4 mg/L to 287.9 mg/L; under Option 3, it decreases 1.6 mg/L to 287.7 mg/L; under Option 4 it decreases 1.5 mg/L to 287.8 mg/L. These numbers represent the national average concentration reduction and do not provide information on the geospatial variability in TSS concentration reduction, which is high.

Table 6-21: Distribution of TSS Concentration based on SPARROW Output for 31,927 RF1 Reaches Receiving Construction Sediment Discharges

Policy Option	Reach Count	Reach Miles	Percent of total construction acres, '92-'01	Average TSS (mg/L)[1,2]	Reduction in Average TSS	Distribution of TSS Concentrations, by percentile (mg/L)[1,2]				
						5th	25th	50th	75th	95th
Baseline	31,927	412,062	93.05%	956.4	–	29.7	109.9	289.3	880.0	6,157.8
Option 1	31,927	412,062	93.05%	955.0	1.5	29.1	109.2	288.6	878.5	6,157.8
Option 2	31,927	412,062	93.05%	954.5	2.0	28.8	108.9	287.9	878.3	6,157.8
Option 3	31,927	412,062	93.05%	954.2	2.2	28.7	108.8	287.7	878.0	6,156.8
Option 4	31,927	412,062	93.05%	954.4	2.1	28.7	108.8	287.8	878.3	6,157.8

[1]Reach-length weighted.
[2]Any TSS concentration above the 95th percentile is assumed to be an outlier and was replaced with the 95th percentile value.

Table 6-22 through *Table 6-26* detail TSS concentrations by EPA region under baseline conditions and for each of the four regulatory options considered. The highest average TSS concentrations are found in Regions 6, 7, 8, and 9 which have average reach-length weighted TSS concentrations greater than 1,400 mg/L under all scenarios. Median TSS concentrations in these regions—ranging from 434.1 to 1,131.2 mg/L under baseline conditions—are also higher than median concentrations in Regions 1, 2, 3, 4, 5, and 10 where the highest median TSS concentration is 196.1 mg/L. Under Option 1, TSS concentrations decline a small amount (by less than 1 percent) from baseline concentrations. TSS concentrations are further reduced under Option 2 and again under Option 3. Under Option 4, concentrations are reduced relative to baseline and concentrations generally fall between those under Option 2 and Option 3. The distribution of concentrations among regions is consistent with the baseline under each of the options.

Table 6-22: Summary of Baseline TSS Concentration in RF1 Reaches Receiving Construction Sediment Discharges, by EPA Region

EPA Region	Reach Count	Reach Miles	Average TSS (mg/L)[1,2]	Distribution of TSS Concentrations, by percentile (mg/L)[1,2]				
				5th	25th	50th	75th	95th
1	1,213	10,740	77.5	9.8	25.2	41.2	57.2	118.7
2	1,111	12,560	110.2	10.5	38.8	79.0	135.7	283.1
3	2,012	25,471	180.6	44.5	83.6	141.7	228.3	447.4
4	7,564	83,058	236.8	33.2	91.4	180.6	311.1	596.8
5	3,918	56,909	296.0	20.6	60.9	196.1	430.4	880.4
6	4,634	70,687	1,567.7	82.9	259.5	641.6	2,137.4	6,157.8
7	3,063	46,903	1,800.8	177.6	484.7	1,050.3	2,008.6	6,157.8
8	4,300	57,513	2,024.6	79.2	359.7	1,131.2	3,104.9	6,157.8
9	1,439	22,280	1,486.7	31.6	157.6	434.1	1,932.3	6,157.8
10	2,673	25,940	239.6	20.6	52.6	108.2	233.2	882.4
Nation	**31,927**	**412,062**	**956.4**	**29.7**	**109.9**	**289.3**	**880.0**	**6,157.8**

[1]Reach-length weighted.
[2]Any TSS concentration above the 95th percentile is assumed to be an outlier and was replaced with the 95th percentile value.

Table 6-23: Summary of Option 1 TSS Concentration in RF1 Reaches Receiving Construction Sediment Discharges, by EPA Region

EPA Region	Reach Count	Reach Miles	Average TSS (mg/L)[1,2]	Distribution of TSS Concentrations, by percentile (mg/L)[1,2]				
				5th	25th	50th	75th	95th
1	1,213	10,740	77.5	9.8	25.2	41.2	57.1	118.7
2	1,111	12,560	110.2	10.5	38.8	79.0	135.6	282.7
3	2,012	25,471	180.5	44.5	83.6	141.5	228.3	447.4
4	7,564	83,058	236.2	32.9	90.9	179.9	311.0	593.7
5	3,918	56,909	295.9	20.6	60.9	196.0	430.4	880.4
6	4,634	70,687	1,564.7	78.8	256.8	639.6	2,130.3	6,157.8
7	3,063	46,903	1,798.8	175.9	482.6	1,048.4	2,007.4	6,157.8
8	4,300	57,513	2,017.8	75.4	354.5	1,126.6	3,104.2	6,157.8
9	1,439	22,280	1,481.5	29.8	153.0	430.7	1,902.6	6,157.8
10	2,673	25,940	239.1	20.3	52.1	107.7	233.2	882.4
Nation	**31,927**	**412,062**	**955.0**	**29.1**	**109.2**	**288.6**	**878.5**	**6,157.8**

[1]Reach-length weighted.
[2]Any TSS concentration above the 95th percentile is assumed to be an outlier and was replaced with the 95th percentile value.

Table 6-24: Summary of Option 2 TSS Concentration in RF1 Reaches Receiving Construction Sediment Discharges, by EPA Region

EPA Region	Reach Count	Reach Miles	Average TSS (mg/L)[1,2]	Distribution of TSS Concentrations, by percentile (mg/L)[1,2]				
				5th	25th	50th	75th	95th
1	1,213	10,740	77.4	9.7	25.2	41.1	57.1	118.7
2	1,111	12,560	110.2	10.5	38.8	79.0	135.5	282.2
3	2,012	25,471	180.4	44.5	83.6	141.5	228.3	447.3
4	7,564	83,058	235.7	32.3	90.7	179.0	310.7	592.9
5	3,918	56,909	295.9	20.6	60.9	195.9	430.4	880.3
6	4,634	70,687	1,563.2	76.2	254.4	637.3	2,128.6	6,157.8
7	3,063	46,903	1,798.4	175.8	482.6	1,048.4	2,007.3	6,157.8
8	4,300	57,513	2,017.7	75.4	354.5	1,126.6	3,104.2	6,157.8
9	1,439	22,280	1,481.2	29.8	152.6	430.3	1,901.1	6,157.8
10	2,673	25,940	238.8	20.0	51.6	107.5	233.2	882.4
Nation	**31,927**	**412,062**	**954.5**	**28.8**	**108.9**	**287.9**	**878.3**	**6,157.8**

[1]Reach-length weighted.
[2]Any TSS concentration above the 95th percentile is assumed to be an outlier and was replaced with the 95th percentile value.

Table 6-25: Summary of Option 3 TSS Concentration in RF1 Reaches Receiving Construction Sediment Discharges, by EPA Region

EPA Region	Reach Count	Reach Miles	Average TSS (mg/L)[1,2]	Distribution of TSS Concentrations, by percentile (mg/L)[1,2]				
				5th	25th	50th	75th	95th
1	1,213	10,740	77.3	9.7	25.2	41.1	57.1	118.6
2	1,111	12,560	110.2	10.5	38.8	79.0	135.5	281.9
3	2,012	25,471	180.4	44.4	83.6	141.5	228.2	447.3
4	7,564	83,058	235.5	32.3	90.5	178.9	310.2	592.2
5	3,918	56,909	295.8	20.6	60.9	195.9	430.4	880.2
6	4,634	70,687	1,562.5	75.4	254.2	637.0	2,124.3	6,156.8
7	3,063	46,903	1,798.1	175.8	481.7	1,048.4	2,007.3	6,156.8
8	4,300	57,513	2,017.6	75.4	354.5	1,126.6	3,104.2	6,156.8
9	1,439	22,280	1,481.0	29.8	152.6	430.3	1,901.0	6,156.8
10	2,673	25,940	238.6	20.0	51.4	107.5	233.2	882.4
Nation	**31,927**	**412,062**	**954.2**	**28.7**	**108.8**	**287.7**	**878.0**	**6,156.8**

[1]Reach-length weighted.
[2]Any TSS concentration above the 95th percentile is assumed to be an outlier and was replaced with the 95th percentile value.

Table 6-26: Summary of Option 4 TSS Concentration in RF1 Reaches Receiving Construction Sediment Discharges, by EPA Region

EPA Region	Reach Count	Reach Miles	Average TSS (mg/L)[1,2]	Distribution of TSS Concentrations, by percentile (mg/L)[1,2]				
				5th	25th	50th	75th	95th
1	1,213	10,740	77.4	9.7	25.2	41.1	57.1	118.7
2	1,111	12,560	110.2	10.5	38.8	79.0	135.5	282.2
3	2,012	25,471	180.4	44.5	83.6	141.5	228.3	447.3
4	7,564	83,058	235.6	32.3	90.7	179.0	310.3	592.5
5	3,918	56,909	295.8	20.6	60.9	195.9	430.4	880.3
6	4,634	70,687	1,562.8	75.6	254.2	637.3	2,127.0	6,157.8
7	3,063	46,903	1,798.3	175.8	481.8	1,048.4	2,007.3	6,157.8
8	4,300	57,513	2,017.7	75.4	354.5	1,126.6	3,104.2	6,157.8
9	1,439	22,280	1,481.2	29.8	152.6	430.3	1,901.1	6,157.8
10	2,673	25,940	238.9	20.0	51.8	107.5	233.2	882.4
Nation	**31,927**	**412,062**	**954.4**	**28.7**	**108.8**	**287.8**	**878.3**	**6,157.8**

[1]Reach-length weighted.
[2]Any TSS concentration above the 95th percentile is assumed to be an outlier and was replaced with the 95th percentile value.

The following tables describe the number of RF1 reach miles improved and the distribution of TSS concentration reductions for each of the four options analyzed for this regulation. As shown in *Table 6-27,* which summarizes the effects of sediment reduction at the national level, Options 1, 2 and 3 improve 94.7 percent of modeled RF1 reach miles receiving construction sediment loading. Options 4 is predicted to lead to improvements in 99.9 percent of RF1 reach miles that receive direct construction loadings. Under Option 1, approximately 93 percent of the improved reach miles show reductions in TSS concentrations of greater than zero but less than 1 mg/L. Under Options 2 through 4, approximately 84 to 86 percent of the improved reach miles are predicted to see a decrease of less than 1 mg/L. While under Option 1, only about 6 percent of improved reach miles are expected see a reduction of 1 to 5 mg/L, under Options 2 through 4, 11 to 12 percent of improved reach miles are expected to improve by 1 to 5 mg/L. Reductions greater than 50 mg/L are expected in less than or equal 0.2 percent of improved miles under all options.

As discussed at the start of *Section 6.6*, the reduction in loadings and concentrations in reaches that receive construction sediment discharge directly also translate into decreased concentrations in downstream reaches. The total number of reach miles that are improved under the four options analyzed is therefore greater than presented in the tables within this chapter. When considering all RF1 reaches, including those that do not receive construction sediment discharge directly, the results show a total of 431,074 reach miles improving under Options 1, 2, and 4. These miles account for 68.7 percent of the total RF1 reach network. Option 3 has an even broader effect, reducing TSS concentrations in a total of 472,402 reach miles (75.3 percent of the RF1 reach network).

Table 6-28 summarizes Option 1's sediment reductions at the regional level. Nationwide, 93 percent of improved RF1 reach miles under Option 1 experience TSS concentration reductions less than 1 mg/L. While all but Region 2 will experience reduction of 1 to 5 mg/L, the number of regions experiencing reductions decreases at higher reduction ranges. All but Regions 2, 3, and 5 will experience reductions between 5 and 10 mg/L, encompassing approximately 0.8 percent of all improved reaches nationally. All but Regions 1, 2, 3, and 5 are predicted to see some reductions between 10 and 50 mg/L encompassing 0.5 percent of all improved reaches nationally. Only four regions (4, 6, 7, and 10) will experience reductions in TSS concentrations greater than 50 mg/L, with a national total of 0.1 percent of improved reaches.

As shown in *Table 6-29,* Option 2 is predicted to result in the same number of total improved reach miles (390,374 miles), but the magnitude of reductions in TSS concentrations in these miles is expected to be greater. All regions are predicted to see a reduction in TSS concentrations of up to 5 mg/L in some reach miles. Approximately 86 and 11 percent of all reach miles improved will experience up to 1 and 5 mg/L reductions, respectively. Some reach miles in all but Region 2 are predicted to improve by 5 to 10 mg/L with approximately 1.6 percent of reach miles improved across the nation in this range. Approximately 1.2 percent of all reach miles improved will see a decrease of 10 to 50 mg/L. Regions 2, 3, and 5 are the only regions without reductions in this range. Nationally, 0.2 percent of all improved reach miles will experience a reduction of greater than 50 mg/L under Option 2. Among regions, Region 6 is predicted to have the greatest number of reach miles with reductions greater than 50 mg/L, followed by Regions 4, 10, 7, and 8.

Option 3 (*Table 6-30*) is estimated to reduce TSS concentrations by more than Option 2 with an additional 21,266 miles of improved rivers and generally a greater magnitude of reductions across the reduction ranges and across regions. Reductions of up to 1 mg/L and 5 mg/L are expected for all regions with a national total of 84 and 12 percent of improved reach miles, respectively. Some reach miles are expected to improve by 5 to 10 mg/L in all regions except Region 2, encompassing 2.0 percent of all improved reach miles nationally. All regions but Regions 2, 3, and 5 are expected to experience improvement in some reach miles by 10 to 50 mg/L including 1.5 percent of all improved reach miles nationally. Greater than 50 mg/L reductions in TSS concentrations are expected in 0.2 percent of all improved reach miles nationally with improvements in Regions 4 and 7 through 10.

Option 4 (*Table 6-31*) is estimated to reduce TSS concentrations in the same number of reach miles as Options 1 and 2, but the reductions are of slightly greater magnitude. Therefore, under Option 4, there is a slightly smaller percentage of total miles improved by up to 1 mg/L and a larger percentage for all other ranges. The spatial distribution of reductions, however, is the same as Option 3, with respect to the regions where reductions are predicted in each of the reduction ranges. Approximately 11.9 and 1.8 percent of improved reach miles are expected to see 1 to 5 mg/L and 5 to 10 mg/L reductions, respectively. Reductions of 10 mg/L to 50 mg/L are expected in 5,414 miles or 1.4 percent of all improved reach miles, and 0.2 percent of improved reach miles are expected to improve by greater than 50 mg/L under Option 4.

Table 6-27: Total RF1 Miles that Receive Direct Construction Loadings with Improvements in TSS Concentrations

Policy Option	Total RF1 Miles	Total Miles with TSS Reductions[1]	Percent of Total Reach Miles with TSS Reductions[1]	Range of Reductions in TSS Concentrations (mg/L)									
				0 < Δ TSS < 1		1 ≤ Δ TSS < 5		5 ≤ Δ TSS < 10		10 ≤ Δ TSS < 50		50 ≤ Δ TSS	
				Reach Miles	% of Total Improved Miles	Reach Miles	% of Total Improved Miles	Reach Miles	% of Total Improved Miles	Reach Miles	% of Total Improved Miles	Reach Miles	% of Total Improved Miles
Option 1	412,062	390,374	94.7%	361,569	92.6%	23,624	6.1%	2,994	0.8%	1,832	0.5%	354	0.1%
Option 2	412,062	390,374	94.7%	336,168	86.1%	42,481	10.9%	6,382	1.6%	4,738	1.2%	605	0.2%
Option 3	412,062	411,640	99.9%	346,379	84.2%	50,221	12.2%	8,084	2.0%	6,147	1.5%	808	0.2%
Option 4	412,062	390,374	94.7%	330,850	84.8%	46,339	11.9%	7,029	1.8%	5,414	1.4%	742	0.2%

[1] Not included in the total are 40 reaches (422 reach miles) for which SPARROW does not predict concentrations. The 99.9% fraction represents all reach miles over which SPARROW calculates TSS concentrations.

Table 6-28: Total RF1 Miles that Receive Direct Construction Loadings with Improvements in TSS Concentrations from Option 1, by EPA Region

EPA Region	Total Improved Reach Miles[1]	Range of Reductions in TSS Concentration (mg/L)									
		0 < Δ TSS < 1		1 ≤ Δ TSS < 5		5 ≤ Δ TSS < 10		10 ≤ Δ TSS < 50		50 ≤ Δ TSS	
		Reach Miles	% of Total	Reach Miles	% of Total	Reach Miles	% of Total	Reach Miles	% of Total	Reach Miles	% of Total
1	10,740	10,610	98.8%	110	1.0%	20	0.2%	0	0.0%	0	0.0%
2	12,554	12,554	100.0%	0	0.0%	0	0.0%	0	0.0%	0	0.0%
3	25,471	25,164	98.8%	307	1.2%	0	0.0%	0	0.0%	0	0.0%
4	82,955	76,258	91.9%	5,657	6.8%	633	0.8%	297	0.4%	111	0.1%
5	56,906	56,474	99.2%	432	0.8%	0	0.0%	0	0.0%	0	0.0%
6	64,954	50,162	77.2%	11,569	17.8%	1,751	2.7%	1,249	1.9%	224	0.3%
7	41,102	37,682	91.7%	3,040	7.4%	257	0.6%	109	0.3%	14	<0.1%
8	49,967	49,350	98.8%	570	1.1%	25	0.1%	21	<0.1%	0	0.0%
9	19,853	18,943	95.4%	778	3.9%	78	0.4%	54	0.3%	0	0.0%
10	25,873	24,374	94.2%	1,161	4.5%	230	0.9%	102	0.4%	6	<0.1%
Nation	**390,374**	**361,569**	**92.6%**	**23,624**	**6.1%**	**2,994**	**0.8%**	**1,832**	**0.5%**	**354**	**0.1%**

[1] Total does not include 40 freshwater reaches (422 reach miles) for which SPARROW does not predict concentrations.

Table 6-29: Total RF1 Miles that Receive Direct Construction Loadings with Improvements in TSS Concentrations from Option 2, by EPA Region

EPA Region	Total Improved Reach Miles[1]	Range of Reductions in TSS Concentration (mg/L)									
		0 < Δ TSS < 1		1 ≤ Δ TSS < 5		5 ≤ Δ TSS < 10		10 ≤ Δ TSS < 50		50 ≤ Δ TSS	
		Reach Miles	% of Total Improved Miles	Reach Miles	% of Total Improved Miles	Reach Miles	% of Total Improved Miles	Reach Miles	% of Total Improved Miles	Reach Miles	% of Total Improved Miles
1	10,740	10,564	98.4%	128	1.2%	29	0.3%	20	0.2%	0	0.0%
2	12,554	12,542	99.9%	12	0.1%	0	0.0%	0	0.0%	0	0.0%
3	25,471	24,751	97.2%	691	2.7%	30	0.1%	0	0.0%	0	0.0%
4	82,955	69,708	84.0%	10,787	13.0%	1,396	1.7%	903	1.1%	161	0.2%
5	56,906	55,616	97.7%	1,236	2.2%	54	0.1%	0	0.0%	0	0.0%
6	64,954	39,267	60.5%	18,668	28.7%	3,685	5.7%	2,970	4.6%	364	0.6%
7	41,102	33,876	82.4%	6,323	15.4%	495	1.2%	394	1.0%	14	<0.1%
8	49,967	48,047	96.2%	1,770	3.5%	104	0.2%	37	0.1%	10	<0.1%
9	19,853	18,041	90.9%	1,515	7.6%	165	0.8%	132	0.7%	0	0.0%
10	25,873	23,758	91.8%	1,352	5.2%	425	1.6%	282	1.1%	56	0.2%
Nation	**390,374**	**336,168**	**86.1%**	**42,481**	**10.9%**	**6,382**	**1.6%**	**4,738**	**1.2%**	**605**	**0.2%**

[1] Total does not include 40 freshwater reaches (422 reach miles) for which SPARROW does not predict concentrations.

Table 6-30: Total RF1 Miles that Receive Direct Construction Loadings with Improvements in TSS Concentrations from Option 3, by EPA Region

EPA Region	Total Improved Reach Miles[1]	Range of Reductions in TSS Concentration (mg/L)									
		0 < Δ TSS < 1		1 ≤ Δ TSS < 5		5 ≤ Δ TSS < 10		10 ≤ Δ TSS < 50		50 ≤ Δ TSS	
		Reach Miles	% of Total Improved Miles	Reach Miles	% of Total Improved Miles	Reach Miles	% of Total Improved Miles	Reach Miles	% of Total Improved Miles	Reach Miles	% of Total Improved Miles
1	10,740	10,538	98.1%	120	1.1%	63	0.6%	20	0.2%	0	0.0%
2	12,560	12,501	99.5%	59	0.5%	0	0.0%	0	0.0%	0	0.0%
3	25,471	24,493	96.2%	908	3.6%	70	0.3%	0	0.0%	0	0.0%
4	83,028	66,798	80.5%	12,850	15.5%	2,021	2.4%	1,186	1.4%	173	0.2%
5	56,906	55,180	97.0%	1,635	2.9%	91	0.2%	0	0.0%	0	0.0%
6	70,630	40,665	57.6%	21,400	30.3%	4,178	5.9%	3,845	5.4%	543	0.8%
7	46,859	37,950	81.0%	7,724	16.5%	667	1.4%	504	1.1%	14	<0.1%
8	57,322	54,799	95.6%	2,273	4.0%	203	0.4%	37	0.1%	10	<0.1%
9	22,204	19,952	89.9%	1,844	8.3%	227	1.0%	169	0.8%	12	0.1%
10	25,920	23,503	90.7%	1,408	5.4%	565	2.2%	388	1.5%	56	0.2%
Nation	**411,640**	**346,379**	**84.2%**	**50,221**	**12.2%**	**8,084**	**2.0%**	**6,147**	**1.5%**	**808**	**0.2%**

[1] Total does not include 40 freshwater reaches (422 reach miles) for which SPARROW does not predict concentrations.

Table 6-31: Total RF1 Miles that Receive Direct Construction Loadings with Improvements in TSS Concentrations from Option 4, by EPA Region

		Range of Reduction in TSS Concentration (mg/L)									
		0 < Δ TSS < 1		1 ≤ Δ TSS < 5		5 ≤ Δ TSS < 10		10 ≤ Δ TSS < 50		50 ≤ Δ TSS	
EPA Region	Total Improved Reach Miles[1]	Reach Miles	% of Total Improved Miles	Reach Miles	% of Total Improved Miles	Reach Miles	% of Total Improved Miles	Reach Miles	% of Total Improved Miles	Reach Miles	% of Total Improved Miles
1	10,740	10,581	98.5%	136	1.3%	4	<0.1%	20	0.2%	0	0.0%
2	12,554	12,546	99.9%	8	0.1%	0	0.0%	0	0.0%	0	0.0%
3	25,471	24,662	96.8%	779	3.1%	30	0.1%	0	0.0%	0	0.0%
4	82,955	68,280	82.3%	11,758	14.2%	1,752	2.1%	991	1.2%	173	0.2%
5	56,906	55,617	97.7%	1,235	2.2%	54	0.1%	0	0.0%	0	0.0%
6	64,954	36,432	56.1%	20,548	31.6%	3,927	6.1%	3,552	5.5%	495	0.8%
7	41,102	33,170	80.7%	6,887	16.8%	568	1.4%	462	1.1%	14	<0.1%
8	49,967	47,677	95.4%	2,048	4.1%	196	0.4%	37	0.1%	10	<0.1%
9	19,853	17,940	90.4%	1,610	8.1%	149	0.8%	142	0.7%	12	0.1%
10	25,873	23,944	92.6%	1,329	5.1%	351	1.4%	211	0.8%	37	0.1%
Nation	**390,374**	**330,850**	**84.8%**	**46,339**	**11.9%**	**7,029**	**1.8%**	**5,414**	**1.4%**	**742**	**0.2%**

[1] Total does not include 40 freshwater reaches (422 reach miles) for which SPARROW does not predict concentrations.

6.6.4 Impacts on Reservoir Sedimentation

Construction discharges affect not only sediment concentrations in receiving water bodies, but they also affect the amount of sedimentation that takes place in reservoirs. As described in *Section 6.2*, reservoirs provide major sites for sediment attenuation within the hydrographic network. This attenuation is estimated in SPARROW for more than 4,000 reservoirs distributed throughout the RF1 network. The model estimates that more than 444 million cubic yards accumulate under baseline conditions. The accumulating sediment may hinder reservoir functions and need to be periodically dredged, as discussed in greater detail in *Chapter 2*. The policy options reduce this amount to various degrees. As shown in *Table 6-32*, Option 1 reduces the amount of sediment accumulating in reservoirs slightly as compared to current conditions (by 558,800 cubic yards or approximately 0.1 percent). Most of this reduction is concentrated in Regions 4, 6, and 7. Options 2, 3, and 4 show greater reductions in reservoir sedimentation, with estimated reductions of 1.16, 1.45, and 1.32 million cubic yards, respectively (or 0.3 percent). They also show more geographically distributed effects, with significant reductions observed in Regions 4, 5, 6, 7, 9, and 10.

Table 6-32: Sediment Accumulation in Reservoirs by Policy Option and EPA Region

EPA Region	Sediment Accumulation (thousand cubic yards)					Reduction in Sediment Accumulation (thousand cubic yards)			
	Baseline	Option 1	Option 2	Option 3	Option 4	Option 1	Option 2	Option 3	Option 4
1	1,278	1,277	1,275	1,275	1,276	1.4	2.9	3.6	2.1
2	2,725	2,725	2,724	2,724	2,724	0.5	1.1	1.4	1.0
3	5,338	5,335	5,332	5,331	5,332	2.6	5.3	6.6	5.5
4	45,667	45,568	45,462	45,411	45,440	98.8	205.1	255.7	226.5
5	35,940	35,933	35,924	35,920	35,923	7.9	16.3	20.3	17.2
6	128,381	127,973	127,530	127,320	127,397	408.5	851.2	1,061.7	984.4
7	42,238	42,213	42,186	42,173	42,179	24.8	51.6	64.3	59.1
8	126,435	126,433	126,430	126,429	126,430	2.2	4.5	5.6	4.7
9	41,860	41,855	41,850	41,847	41,849	4.8	10.0	12.4	10.5
10	14,744	14,737	14,729	14,726	14,732	7.3	14.8	18.4	11.8
Nation	**444,606**	**444,047**	**443,443**	**443,156**	**443,283**	**558.8**	**1,162.7**	**1,449.9**	**1,322.9**

6.6.5 Estimated Changes in Nutrient Concentrations for all RF1 Watersheds Receiving Loading from Construction Sites

Table 6-33 and *Table 6-34* summarize national average reach-length weighted TN and TP concentrations and their distribution under the baseline conditions and the four regulatory options (see *Chapter 1* for option descriptions) for reaches receiving construction sediment discharges. TN and TP concentrations in the RF1 network vary over a wide range. Nutrient criteria established by EPA for rivers and streams vary across ecoregions from 0.12 mg/L to 2.18 mg/L for TN and from 10 µg/L to 128 µg/L for TP (USEPA 2009d).

Estimated TN concentrations under baseline conditions range between 0.37 and 14.95 mg/L for the 5th and 95th percentiles, respectively. Median (50th percentile) TN concentrations are approximately 1.65 mg/L under the baseline scenario. Under Options 1 and 2, the median concentration remains relatively unchanged at 1.65 mg/L; under Options 3 and 4, it decreases 0.01 mg/L to 1.64 mg/L. Overall, the largest reduction in TN concentrations is predicted under Option 3 with a 0.5 percent reduction in median TN concentrations relative to baseline levels. These numbers represent the national average concentration reduction, however, and do not reflect the wide geospatial variability in TN concentration reductions.

Table 6-33: Distribution of TN Concentration Based on SPARROW Output for 31,927 RF1 Reaches Receiving Construction Sediment Discharges

Policy Option	Reach Count	Reach Miles	Average TN (mg/L)[1,2]	Reduction in Average TN	Distribution of TN Concentrations, by Percentile (mg/L)[1,2]				
					5th	25th	50th	75th	95th
Baseline	31,927	412,062	18.22	–	0.37	0.91	1.65	3.88	14.95
Option 1	31,927	412,062	18.21	0.01	0.37	0.91	1.65	3.87	14.91
Option 2	31,927	412,062	18.21	0.02	0.36	0.91	1.65	3.87	14.91
Option 3	31,927	412,062	18.21	0.02	0.36	0.90	1.64	3.87	14.91
Option 4	31,927	412,062	18.21	0.02	0.36	0.91	1.65	3.87	14.91

[1]Reach-length weighted.
[2]Nutrient reductions are based on TSS reductions. Any TSS concentration above the 95th percentile is assumed to be an outlier and was replaced with the 95th percentile value.

Values for TP concentrations for the 5th and 95th percentile bounds under baseline conditions are 30.5 and 2,907.3 µg/L, respectively. Median (50th percentile) TP concentrations are approximately 253.2 µg/L under the baseline scenario. Under Option 1, the median concentration decreases 1.4 µg/L to 251.8 µg/L. The median concentration under Option 2 decreases 1.9 µg/L to 251.3 µg/L and under Option 3 and 4, it decreases 2.1 µg/L to 251.1 µg/L. These numbers represent the national average concentration reduction, however, and do not reflect the wide geospatial variability in TP concentration reductions.

Table 6-34: Distribution of TP Concentration Based on SPARROW Output for 31,927 RF1 Reaches Receiving Construction Sediment Discharges

Policy Option	Reach Count	Reach Miles	Average TP (µg/L)[1,2]	Reduction in Average TP	Distribution of TP Concentrations, by Percentile (µg/L)[1,2]				
					5th	25th	50th	75th	95th
Baseline	31,927	412,062	3,921.6		30.5	104.4	253.2	685.3	2,907.3
Option 1	31,927	412,062	3,916.3	5.3	29.9	103.8	251.8	683.7	2,886.4
Option 2	31,927	412,062	3,915.9	5.7	29.6	103.4	251.3	683.6	2,886.4
Option 3	31,927	412,062	3,915.7	5.9	29.6	103.3	251.1	683.5	2,885.1
Option 4	31,927	412,062	3,915.9	5.8	29.6	103.4	251.1	683.5	2,886.4

[1]Reach-length weighted.
[2] Nutrient reductions are based on TSS reductions. Any TSS concentration above the 95th percentile is assumed to be an outlier and was replaced with the 95th percentile value.

Changes in nutrient concentrations are based on in-stream sediment reductions, as described in *Section 6.4.* Therefore, the distribution and magnitude of percent change in reach miles will be the same as for TSS, as presented in *Section 6.6.3.* These tables are not recreated below for TN and TP.

Table 6-35 through *Table 6-39* detail TN concentrations by EPA region under baseline conditions and for each of the four regulatory options considered. The highest average TN concentrations are found in Regions 8 and 9, which have average reach-length weighted TN concentrations between 40.01 and 46.56 mg/L under all scenarios. However, the highest median TN concentrations are found in Regions 5 and 7, with concentrations of 3.68 and 5.11 mg/L, respectively. Regions 8 and 9 appear to have a skewed distribution with a smaller portion of reach miles at high concentrations skewing the average higher than other regions. Under all four options, TN concentrations decline a small amount, approximately 0.1 percent, from baseline concentrations.

Table 6-35: Summary of Baseline TN Concentration in RF1 Reaches Receiving Construction Sediment Discharges, by EPA Region

EPA Region	Reach Count	Reach Miles	Average TN (mg/L)[1]	Distribution of TN Concentrations, by Percentile (mg/L)[1]				
				5th	25th	50th	75th	95th
1	1,213	10,740	1.32	0.24	0.59	0.85	1.27	2.57
2	1,111	12,560	19.43	0.33	0.95	1.37	1.98	3.85
3	2,012	25,471	1.88	0.61	1.02	1.35	1.95	4.35
4	7,564	83,058	4.39	0.29	0.70	1.06	1.59	3.06
5	3,918	56,909	7.76	0.51	1.49	3.68	7.50	15.86
6	4,634	70,687	26.15	0.45	1.07	1.83	3.26	12.38
7	3,063	46,903	22.87	1.10	2.64	5.11	9.56	21.50
8	4,300	57,513	40.01	0.50	1.08	2.44	5.99	28.66
9	1,439	22,280	46.56	0.24	0.55	1.28	3.35	54.43
10	2,673	25,940	5.30	0.23	0.49	0.83	1.58	5.29
Nation	**31,927**	**412,062**	**18.22**	**0.37**	**0.91**	**1.65**	**3.88**	**14.95**

[1]Reach-length weighted.

Table 6-36: Summary Option 1 TN Concentration in RF1 Reaches Receiving Construction Sediment Discharges, by EPA Region

EPA Region	Reach Count	Reach Miles	Average TN (mg/L)[1,2]	Distribution of TN Concentrations, by Percentile (mg/L)[1,2]				
				5th	25th	50th	75th	95th
1	1,213	10,740	1.32	0.24	0.59	0.85	1.27	2.57
2	1,111	12,560	19.43	0.33	0.95	1.37	1.98	3.85
3	2,012	25,471	1.88	0.61	1.02	1.35	1.95	4.35
4	7,564	83,058	4.39	0.28	0.70	1.06	1.58	3.06
5	3,918	56,909	7.76	0.51	1.49	3.68	7.49	15.86
6	4,634	70,687	26.12	0.43	1.06	1.83	3.24	12.37
7	3,063	46,903	22.86	1.08	2.62	5.10	9.53	21.50
8	4,300	57,513	39.99	0.48	1.07	2.42	5.96	28.64
9	1,439	22,280	46.51	0.23	0.54	1.28	3.26	54.43
10	2,673	25,940	5.29	0.23	0.49	0.83	1.58	5.29
Nation	**31,927**	**412,062**	**18.21**	**0.36**	**0.91**	**1.65**	**3.87**	**14.91**

[1]Reach-length weighted.
[2] Nutrient reductions are based on TSS reductions. Any TSS concentration above the 95th percentile is assumed to be an outlier and was replaced with the 95th percentile value.

Table 6-37: Summary of Option 2 TN Concentration in RF1 Reaches Receiving Construction Sediment Discharges, by EPA Region

EPA Region	Reach Count	Reach Miles	Average TN (mg/L)[1,2]	Distribution of TN Concentrations, by Percentile (mg/L)[1,2]				
				5th	25th	50th	75th	95th
1	1,213	10,740	1.32	0.24	0.59	0.85	1.27	2.57
2	1,111	12,560	19.43	0.33	0.95	1.37	1.98	3.85
3	2,012	25,471	1.88	0.61	1.02	1.35	1.95	4.34
4	7,564	83,058	4.38	0.28	0.70	1.06	1.58	3.06
5	3,918	56,909	7.76	0.51	1.48	3.68	7.49	15.86
6	4,634	70,687	26.12	0.43	1.06	1.83	3.23	12.35
7	3,063	46,903	22.85	1.08	2.62	5.10	9.49	21.50
8	4,300	57,513	39.99	0.48	1.07	2.42	5.96	28.64
9	1,439	22,280	46.51	0.23	0.54	1.28	3.26	54.43
10	2,673	25,940	5.29	0.23	0.49	0.83	1.58	5.29
Nation	**31,927**	**412,062**	**18.21**	**0.36**	**0.91**	**1.65**	**3.87**	**14.91**

[1]Reach-length weighted.
[2] Nutrient reductions are based on TSS reductions. Any TSS concentration above the 95th percentile is assumed to be an outlier and was replaced with the 95th percentile value.

Table 6-38: Summary of Option 3 TN Concentration in RF1 Reaches Receiving Construction Sediment Discharges, by EPA Region

EPA Region	Reach Count	Reach Miles	Average TN (mg/L)[1,2]	Distribution of TN Concentrations, by Percentile (mg/L)[1,2]				
				5th	25th	50th	75th	95th
1	1,213	10,740	1.32	0.24	0.59	0.85	1.27	2.57
2	1,111	12,560	19.43	0.33	0.95	1.37	1.98	3.85
3	2,012	25,471	1.88	0.61	1.02	1.35	1.95	4.34
4	7,564	83,058	4.38	0.28	0.70	1.06	1.58	3.06
5	3,918	56,909	7.75	0.51	1.48	3.68	7.49	15.86
6	4,634	70,687	26.11	0.42	1.06	1.83	3.23	12.34
7	3,063	46,903	22.85	1.08	2.61	5.10	9.49	21.50
8	4,300	57,513	39.99	0.48	1.07	2.42	5.96	28.64
9	1,439	22,280	46.51	0.23	0.54	1.28	3.26	54.43
10	2,673	25,940	5.29	0.23	0.49	0.83	1.58	5.29
Nation	**31,927**	**412,062**	**18.21**	**0.36**	**0.90**	**1.64**	**3.87**	**14.91**

[1]Reach-length weighted.
[2] Nutrient reductions are based on TSS reductions. Any TSS concentration above the 95th percentile is assumed to be an outlier and was replaced with the 95th percentile value.

Table 6-39: Summary of Option 4 TN Concentration in RF1 Reaches Receiving Construction Sediment Discharges, by EPA Region

EPA Region	Reach Count	Reach Miles	Average TN (mg/L)[1,2]	Distribution of TN Concentrations, by Percentile (mg/L)[1,2]				
				5th	25th	50th	75th	95th
1	1,213	10,740	1.32	0.24	0.59	0.85	1.27	2.57
2	1,111	12,560	19.43	0.33	0.95	1.37	1.98	3.85
3	2,012	25,471	1.88	0.61	1.02	1.35	1.95	4.34
4	7,564	83,058	4.38	0.28	0.70	1.06	1.58	3.06
5	3,918	56,909	7.76	0.51	1.48	3.68	7.49	15.86
6	4,634	70,687	26.11	0.42	1.06	1.83	3.23	12.34
7	3,063	46,903	22.85	1.08	2.62	5.10	9.49	21.50
8	4,300	57,513	39.99	0.48	1.07	2.42	5.96	28.64
9	1,439	22,280	46.51	0.23	0.54	1.28	3.26	54.43
10	2,673	25,940	5.29	0.23	0.49	0.83	1.58	5.29
Nation	**31,927**	**412,062**	**18.21**	**0.36**	**0.90**	**1.64**	**3.87**	**14.91**

[1]Reach-length weighted.
[2] Nutrient reductions are based on TSS reductions. Any TSS concentration above the 95th percentile is assumed to be an outlier and was replaced with the 95th percentile value.

Table 6-40 through *Table 6-44* detail TP concentrations by EPA region under baseline conditions and for each of the four regulatory options considered. The highest average TP concentrations are found in Regions 2, 6, 7, 8, and 9, which have average reach-length weighted TP concentrations greater than 1,600 µg/L under all scenarios. These regions, with the exception of Region 2, also show the highest median TP concentrations (ranging from 489.0 to 659.0 µg/L under baseline conditions). Option 1 shows the smallest TP concentration reductions, with Options 2 and 3 showing slightly higher reductions. Option 4 shows TP concentration reductions that are between those under Option 2 and Option 3, similar to trends observed for TN.

Table 6-40: Summary of Baseline TP Concentration in RF1 Reaches Receiving Construction Sediment Discharges, by EPA Region

EPA Region	Reach Count	Reach Miles	Average TP (µg/L)[1]	Distribution of TP Concentrations, by Percentile (µg/L)[1]				
				5th	25th	50th	75th	95th
1	1,213	10,740	109.5	9.9	29.6	49.2	86.5	233.8
2	1,111	12,560	1,697.5	16.6	47.2	91.2	152.1	403.0
3	2,012	25,471	209.3	37.7	80.2	135.4	233.5	633.6
4	7,564	83,058	885.6	30.0	81.8	139.5	241.8	547.4
5	3,918	56,909	651.2	18.6	75.6	225.3	475.9	1,246.4
6	4,634	70,687	8,110.6	63.4	208.4	519.5	1,216.9	5,557.8
7	3,063	46,903	3,380.2	140.6	377.8	603.9	1,012.7	2,853.1
8	4,300	57,513	7,282.3	69.0	249.5	659.0	1,449.5	5,803.0
9	1,439	22,280	13,544.9	37.0	114.9	489.0	1,652.3	30,733.9
10	2,673	25,940	965.7	22.1	56.3	113.4	303.0	1,318.7
Nation	**31,927**	**412,062**	**3,921.6**	**30.5**	**104.4**	**253.2**	**685.3**	**2,907.3**

[1]Reach-length weighted.

Table 6-41: Summary of Option 1 TP Concentration in RF1 Reaches Receiving Construction Sediment Discharges, by EPA Region

EPA Region	Reach Count	Reach Miles	Average TP (µg/L)[1,2]	Distribution of TP Concentrations, by Percentile (µg/L)[1,2]				
				5th	25th	50th	75th	95th
1	1,213	10,740	109.4	9.9	29.5	49.2	86.5	233.8
2	1,111	12,560	1,697.5	16.6	47.2	91.2	152.0	403.0
3	2,012	25,471	209.2	37.7	80.1	135.3	233.4	633.6
4	7,564	83,058	885.1	29.8	81.4	139.1	241.1	547.4
5	3,918	56,909	651.2	18.6	75.5	225.3	475.8	1,246.4
6	4,634	70,687	8,096.5	61.3	205.2	517.0	1,213.2	5,557.7
7	3,063	46,903	3,377.4	138.4	375.7	603.3	1,010.4	2,853.1
8	4,300	57,513	7,277.2	66.6	246.4	656.7	1,445.2	5,780.2
9	1,439	22,280	13,513.0	36.2	111.7	476.2	1,644.4	30,733.9
10	2,673	25,940	965.2	21.5	56.1	112.3	303.0	1,314.0
Nation	**31,927**	**412,062**	**3,916.3**	**29.9**	**103.8**	**251.8**	**683.7**	**2,886.4**

[1]Reach-length weighted.
[2] Nutrient reductions are based on TSS reductions. Any TSS concentration above the 95th percentile is assumed to be an outlier and was replaced with the 95th percentile value.

Table 6-42: Summary of Option 2 TP Concentration in RF1 Reaches Receiving Construction Sediment Discharges, by EPA Region

EPA Region	Reach Count	Reach Miles	Average TP (µg/L)[1,2]	Distribution of TP Concentrations, by Percentile (µg/L)[1,2]				
				5th	25th	50th	75th	95th
1	1,213	10,740	109.4	9.9	29.5	49.2	86.5	233.8
2	1,111	12,560	1,697.5	16.6	47.2	91.2	152.0	402.9
3	2,012	25,471	209.1	37.7	80.0	135.2	233.3	633.5
4	7,564	83,058	884.7	29.3	81.2	138.9	240.2	547.3
5	3,918	56,909	651.1	18.6	75.5	225.3	475.8	1,246.4
6	4,634	70,687	8,095.2	59.0	203.8	516.6	1,211.4	5,557.7
7	3,063	46,903	3,377.2	138.4	374.9	602.8	1,008.8	2,853.1
8	4,300	57,513	7,277.2	66.6	246.4	656.6	1,445.2	5,780.1
9	1,439	22,280	13,512.8	35.3	111.4	475.1	1,644.4	30,733.9
10	2,673	25,940	964.9	21.5	55.7	112.3	300.4	1,289.3
Nation	**31,927**	**412,062**	**3,915.9**	**29.6**	**103.4**	**251.3**	**683.6**	**2,886.4**

[1]Reach-length weighted.
[2] Nutrient reductions are based on TSS reductions. Any TSS concentration above the 95th percentile is assumed to be an outlier and was replaced with the 95th percentile value.

Table 6-43: Summary of Option 3 TP Concentration in RF1 Reaches Receiving Construction Sediment Discharges, by EPA Region

EPA Region	Reach Count	Reach Miles	Average TP (µg/L)[1,2]	Distribution of TP Concentrations, by Percentile (µg/L)[1,2]				
				5th	25th	50th	75th	95th
1	1,213	10,740	109.4	9.9	29.4	49.2	86.5	233.7
2	1,111	12,560	1,697.5	16.6	47.2	91.1	152.0	402.9
3	2,012	25,471	209.1	37.7	80.0	135.1	233.3	633.5
4	7,564	83,058	884.5	29.2	81.2	138.6	240.0	547.3
5	3,918	56,909	651.1	18.5	75.5	225.3	475.8	1,246.4
6	4,634	70,687	8,094.6	58.6	203.6	516.2	1,209.8	5,557.7
7	3,063	46,903	3,376.9	138.4	374.9	602.2	1,008.8	2,850.8
8	4,300	57,513	7,277.0	66.6	246.4	656.6	1,445.2	5,780.1
9	1,439	22,280	13,512.7	35.2	111.4	474.5	1,644.4	30,733.9
10	2,673	25,940	964.7	21.5	55.6	112.3	297.9	1,277.5
Nation	**31,927**	**412,062**	**3,915.7**	**29.6**	**103.3**	**251.1**	**683.5**	**2,885.1**

[1]Reach-length weighted.
[2] Nutrient reductions are based on TSS reductions. Any TSS concentration above the 95th percentile is assumed to be an outlier and was replaced with the 95th percentile value.

Table 6-44: Summary of Option 4 TP Concentration in RF1 Reaches Receiving Construction Sediment Discharges, by EPA Region

EPA Region	Reach Count	Reach Miles	Average TP (µg/L)[1,2]	Distribution of TP Concentrations, by Percentile (µg/L)[1,2]				
				5th	25th	50th	75th	95th
1	1,213	10,740	109.4	9.9	29.5	49.2	86.5	233.8
2	1,111	12,560	1,697.5	16.6	47.2	91.2	152.0	402.9
3	2,012	25,471	209.1	37.7	80.0	135.2	233.3	633.5
4	7,564	83,058	884.6	29.3	81.2	138.9	240.2	547.3
5	3,918	56,909	651.1	18.6	75.5	225.3	475.8	1,246.4
6	4,634	70,687	8,094.9	58.8	203.7	516.2	1,211.2	5,557.7
7	3,063	46,903	3,377.1	138.4	374.9	602.4	1,008.8	2,853.1
8	4,300	57,513	7,277.1	66.6	246.4	656.6	1,445.2	5,780.1
9	1,439	22,280	13,512.8	35.3	111.4	475.0	1,644.4	30,733.9
10	2,673	25,940	965.0	21.5	55.8	112.3	301.2	1,297.2
Nation	**31,927**	**412,062**	**3,915.9**	**29.6**	**103.4**	**251.1**	**683.5**	**2,886.4**

[1]Reach-length weighted.
[2] Nutrient reductions are based on TSS reductions. Any TSS concentration above the 95th percentile is assumed to be an outlier and was replaced with the 95th percentile value.

7 Benefits to Navigation

Navigable waterways, including rivers, lakes, bays, shipping channels and harbors, are an integral part of the United States' industrial transportation network. Navigable channels are prone to reduced functionality due to sediment build-up, which can reduce the navigable depth and width of the waterway (Clark et al. 1985). Removing sediment to keep navigable waterways passable requires dredging. The U.S. Army Corps of Engineers (USACE) spends an average of more than $572 million (2008$) every year to dredge waterways and keep them passable (USACE 2009).

Implementation of the regulation is expected to result in less frequent dredging of navigable waterways due to reduced sediment runoff from construction sites. This will result in avoided costs to the government and private entities responsible for maintenance of navigational shipping channels, harbors, rivers, and other waterways. For example, prior studies of benefits of the Conservation Reserve program found that a reduction of 205 million tons of sediment erosion from farmland would result in approximately $526 million (2008$) in benefits to navigation from decreased dredging (Ribaudo 1989). Finally, unless there is overdredging to compensate or sedimentation is monitored so that dredging activity may be timed optimally, water bodies will be on average less navigable even with a dredging program.

This chapter presents EPA's analysis of the navigable waterway maintenance costs that would be avoided by implementation of the regulation. This approach represents an appropriate measure of social benefits for cases in which a policy change reduces costs to producers (in this case "producers" of navigable waterways), but in which price effects are minimal (cf. Boardman et al. 2001, p. 70-74). Additional environmental benefits of reduced dredging are not estimated due to the highly place-specific nature of these benefits and difficulty in estimating these costs over the national scale. Hence, in this regard the Agency's benefit estimate in this chapter is understated. This analysis includes the following steps:

- Identifying navigable waterways that are regularly dredged and estimating the frequency of regular dredging in each waterway
- Estimating the navigable waterway maintenance cost per cubic yard of sediment dredged
- Estimating the total cost of navigable waterway maintenance under the baseline and post-compliance scenarios
- Estimating cost saving from decreased dredging of navigable waterways due to the reduction in sediment runoff from construction sites.

7.1 Data Sources

EPA relied on the USACE Dredging Information System in analyzing benefits to navigation from the regulation. The dredging database catalogs all USACE dredging contracts from 1990 to 2008 and all USACE-conducted dredging jobs from 1995 to 2008, and provides information on the location, dates, cost, and amount of sediment dredged for each dredging job or contract. For the purpose of this analysis, a "dredging occurrence" refers to a single record in these data, containing a location and time of dredging, while a "dredging job" refers to a single location of dredging that may have more than one dredging occurrence, and thus be repeated in the data.

Where the USACE Dredging Information System did not contain cost or quantity of sediment dredged for a listed dredging occurrence, EPA estimated the missing information from other jobs. The Agency calculated 10th, 50th, and 90th percentile costs and quantity boundaries and used these to fill in all incomplete records in that region. EPA included 50th percentile (median) estimates in the summary data presented in *Table 7-1* through *Table 7-4* and in the midpoint estimate of total dredging costs. The 10th and 90th percentile cost and amount dredged estimates were used in the low and high total dredging cost estimates, respectively. For more details, see *Section 7.4.3: Sensitivity Analysis*.

Table 7-1 summarizes dredging information for dredging jobs for the period 1995–2008, including dredging performed by USACE and dredging contracted to other firms. The cost data were adjusted to 2008$ using the construction cost index (CCI) (Construction Cost Index 2008). During this period, USACE funded nearly 3,400 dredging occurrences, worth more than $9 billion (2008$). This dredging removed more than 2.6 billion cubic yards of sediment from more than 1,100 navigable reaches. The region with the most dredging was Region 4, reporting more than 900 dredging occurrences between 1995 and 2008, about 65 per year. Region 6 reported the largest volume of sediment removed, with nearly 1.5 billion cubic yards removed from navigable waterways over this 14-year period. Though benefits are also dependent on the amount of sediment reduction in a region, regions with more frequent dredging are more likely to experience greater benefits from reduced sedimentation in navigable waterways.

Table 7-1: Dredging in U.S. Navigable Waterways, 1995–2008

EPA Region	Number of Dredging Occurrences	Total Sediment Removed (millions of cubic yards)	Total Cost (millions of 2008$)
1	79	3.7	$45.7
2	278	111.4	$801.1
3	291	124.5	$902.8
4	921	540.9	$3,121.3
5	739	112.8	$517.8
6	647	1,483.3	$2,706.4
7	22	17.3	$39.5
8	1	0.5	$2.3
9	129	67.7	$449.3
10	290	153.5	$499.9
Total	**3,397**	**2,615.7**	**$9,086.3**

Source: Dredging Information System (USACE 2009).

Table 7-2 summarizes dredging activity from 1995 through 2008 by type of entity performing dredging work (private or government). As noted above, USACE spent more than $9 billion (2008$) over the past 14 years to fund nearly 3,400 dredging occurrences. The majority of dredging work (58 percent) was contracted out, with USACE performing about 42 percent of the work over this period. The individual costs of these dredging occurrences range considerably, from around $10,000 to more than $100 million, but averaging about $3 million (2008$).

Table 7-2: Dredging of Navigable Waterways Performed or Contracted by USACE, 1995–2008

	Reported Occurrences of Dredging	Percent of Total Reported Dredging Occurrences	Reported Sediment Dredged (millions of cubic yards)	Percent of Total Reported Sediment Dredged	Total Reported Cost (millions of 2008$)	Percent of Total Reported Cost
USACE	1,414	42%	1,067.0	41%	$3,963	44%
Contract	1,983	58%	1,548.7	59%	$5,124	56%
Total	**3,397**	**100%**	**2,615.7**	**100%**	**$9,086**	**100%**

Source: Dredging Information System (USACE 2009).

7.2 Identifying Waterways That Are Dredged, the Frequency of Dredging, and the Quantity of Sediment Dredged

7.2.1 Determining Dredging Job Locations

Location information is given by latitude–longitude coordinates for some jobs, but for others, location information was limited to the name of the job (usually the waterway dredged) and the USACE district that performed the job. To identify the nearest reach segment, EPA used an unprojected version of the ERF 1.2 (Enhanced Reach File) from USGS. For this analysis, EPA researched latitude–longitude coordinates for jobs where they were not provided and linked them to Reach File Version 1.0 (RF1) reaches. Each latitude/longitude of interest was matched to the nearest point in the ERF 1.2 universe of points using a spherical model of the earth and a standard haversine distance formula. To associate the point with an RF1 reach number, EPA used the "RR" (River Reach) attribute from the ERF 1.2 attribute table. No reach types were excluded from consideration in the nearest reach calculation. Because these coordinates may be inexact, dredging activity was aggregated to the state level for the calculations in this analysis and is presented at a regional level.

7.2.2 Identifying the Baseline Frequency of Dredging

Continuous sediment deposition in a navigable waterway is likely to require repeated dredging to maintain navigability. Between 1995 and 2008, 1,272 sites required dredging on at least one occasion, and 551 required dredging more than once. *Table 7-3* shows these statistics for each EPA region. The number of dredging jobs varies by region, and is likely to be influenced by the size of the region, its number of navigable waterways, and their economic importance. Because the data available from USACE reflect a relatively short period of time (14 years) and dredging data show a large degree of variability in the frequency of dredging occurrences, EPA calculated an average frequency of recurrence for each dredging job by dividing 14 years by the number of occurrences of each job over these 14 years.[7] The EPA-estimated regional average recurrence intervals range from 9.6 to 14 years, including jobs that occurred only once in 14 years. For all dredging jobs in the United States, the average frequency of recurrence is about 10 years. Excluding jobs that occurred only once over this period, the regional average recurrence interval ranges from 4.4 to 7 years, averaging slightly less than 5 years at the national level.

7 As described later in this section, EPA used the number of days between dredging occurrences and imposed a minimum number of intervening days for a dredging occurrence to be considered unique. Occurrences separated by fewer than this minimum number of days were considered as a single occurrence.

Prior studies (Clark et al. 1985; Ribaudo 1989) of the avoided costs from reduction in sedimentation of navigable waterways, shipping channels, and harbors assumed a linear relationship between sediment runoff and dredging activity, implying that every ton of sediment entering navigable waterways would eventually be dredged up. Because dredging is an expensive and environmentally disruptive activity, it is unlikely that every navigable waterway that receives sediment will eventually be dredged.

For this analysis, EPA assumed that navigable waterways that are regularly dredged would benefit from reducing the amount of sediment that needs to be dredged up. Although other waterways may benefit from reduced sedimentation, EPA based its estimation of reduced dredging activity based on historic dredging data from USACE. For dredging jobs that occurred only once between 1995 and 2008, EPA assumed that this job or one similar to it will occur over the same time frame in the future. However, these single occurrence jobs are excluded from the low range estimate of the analysis (see *Section 7.4.3: Sensitivity Analysis)*.

Table 7-3: Dredging Jobs and Recurrence Intervals, 1995- 2008

EPA Region	Number of Jobs	Number of Recurring Jobs	Average Interval (years)	Average Interval for Recurring Jobs (years)
1	46	11	11.8	4.6
2	105	48	9.7	4.7
3	133	50	10.7	5.3
4	294	126	10.0	4.6
5	293	139	9.6	4.7
6	226	104	9.6	4.4
7	18	2	13.2	7.0
8	1	0	14.0	N/A
9	45	21	9.6	4.5
10	111	50	9.9	4.9
Total	**1,272**	**551**	**10.0**	**4.7**

Source: Dredging Information System (USACE 2009).

EPA used the USACE dredging data to identify navigable waterways that are dredged (dredging jobs), as well as the frequency at which dredging activity occurs. For each dredging job included in the analysis, the Agency identified:

- The number of occurrences of this dredging job
- The time elapsed between occurrences of dredging.

EPA excluded from this analysis any occurrences where the time elapsed since the last dredging occurrence was less than 30 days due to uncertainty as to whether these occurrences were continuations of an earlier occurrence of dredging. From this information, EPA established a dredging schedule for each recurring dredging job for the next 14 years. For the 721 jobs that only occurred once from 1995 to 2008, the number of future dredging jobs is set to one, except in the low estimate, where it is set to zero (see *Section 7.4.3: Sensitivity Analysis)*. For other dredging jobs, EPA divided 14 years by the number of occurrences of that job between 1995 and 2008 to obtain an average interval of dredging for that job in years.

7.2.3 Identifying the Amount of Sediment Dredged

For most jobs, the USACE data reports sediment dredged (in cubic yards), but in cases where this data was missing, EPA filled in the missing data with percentile values, as described in *Section 7.1*. In addition, for occurrences of the same job that were within 30, 90, or 180[8] days of one another, the occurrences were consolidated into one occurrence and total amount of sediment dredged was summed. The total quantity of sediment dredged for each job (single location) over the past 14 years was divided by the number of occurrences of that job to calculate an average quantity of sediment dredged for an occurrence of that job. EPA assumed that this quantity of sediment would be dredged each time the job occurs in the future under the baseline scenario, and that it would be reduced due to the regulation (see *Section 7.4.1*).

7.3 Estimating the Navigational Maintenance Cost per Cubic Yard of Sediment Removed

Table 7-4 shows the average cost of dredging for each EPA region. Average unit dredging costs vary considerably over these regions, from around $1.80 per cubic yard in Region 6 to nearly $11 per cubic yard in Region 1. The average unit cost of dredging for the entire United States is $3.40 per cubic yard.

Table 7-4: Costs of Recurring Dredging, 1995-2006

EPA Region	Total Cost (millions of 2008$)	Total Sediment Removed (millions of cubic yards)	Average Cost per cubic yard
1	$50.1	4.8	$10.49
2	$1,057.3	147.0	$7.19
3	$1,091.7	157.2	$6.95
4	$3,862.2	705.9	$5.47
5	$625.7	133.4	$4.69
6	$3,443.4	1,915.7	$1.80
7	$48.6	18.6	$2.60
8	$2.3	0.5	$4.49
9	$544.2	84.1	$6.47
10	$717.7	196.7	$3.65
Total	**$11,443.1**	**3,363.9**	**$3.40**

Source: USACE Dredging Information System (2009).

Though USACE does not disaggregate its reported dredging costs, the cost of sediment dredging typically includes the following components (Sohngen and Rausch 1998):

- The cost of dredging sediment from the bed and loading onto the boat
- The cost of transporting the material to a disposal facility
- The cost of confining or disposing of the dredged sediment.

[8] The threshold for determining whether an occurrence was an extension of the previous occurrence varies between the low (180 days), midpoint (90 days), and high (30 days) estimates.

7.4 Estimating the Total Cost of Navigable Waterway Maintenance Under Different Policy Scenarios

Avoided costs of navigational waterway dredging were estimated by comparing estimated future dredging costs under the no policy scenario with estimated future dredging costs under the post-compliance scenario.

7.4.1 Estimating the Reduction in Sediment Dredging in Navigable Waterways Due to the Reduction in Discharge from Construction Sites

EPA estimated changes in sediment deposition in navigable waterways under different regulatory scenarios using output from SPARROW. *Chapter 6* provides a description of SPARROW as well as input data used in this analysis. For the post-compliance scenarios, EPA calculated the percentage change in sediment yield from the baseline for each RF1 reach in the SPARROW analysis. The quantity of sediment for each dredging job (calculated as described in *Section 7.2.3*) was multiplied by this percentage change in sediment yield to obtain a post-compliance quantity of sediment to be dredged each time a job occurs in the next 14 years[9]. If an RF1 reach was not assigned to a dredging job due to data limitations or the reach assigned to that dredging job was not included in the SPARROW model (such as coastal reaches), the state-wide average reduction in sediment yield was applied to that job.

7.4.2 Estimating the Total Cost of Navigable Waterway Maintenance Under the Baseline and Post-Compliance Scenarios

EPA estimated future dredging costs over a period of 14 years from 2010 to 2023 under both the baseline and post-compliance scenarios. For this cost calculation, average cost per cubic yard of sediment dredged, total amount of sediment dredged, and the average interval between dredging jobs were calculated for each relevant job in the USACE Dredging Information System. For the post-compliance scenarios, total amount of sediment dredged was determined as described in *Section 7.4.1*.

Because costs will occur whenever the waterbody is dredged rather than on an annual basis, each reach will have a unique stream of costs derived from its individual dredging frequency, cost, dredging volume, and reduction in sediment. Thus, EPA calculated the present value of dredging costs over the 14-year period using a dredging schedule for each dredging job and then discounted the estimated values using both 3 and 7 percent annual interest rates. EPA estimated the present value of each dredging job as follows:

- ➢ Estimating the average recurrence interval for the job over the 14-year period for each job by dividing 14 years by its number of occurrences in the USACE dredging data. This produces *I* in the equation below.
- ➢ Entering this information along with the cost per cubic yard dredged, number of cubic yards dredged, and percentage of sediment requiring dredging in the post-compliance scenario into the formula below:

$$PV = \sum_{n=1}^{14/I} \left[\frac{Q_b * R_{pc} * C}{(1+d)^{(I*n)}} \right] \quad \text{(Eq. 7-1)}$$

9 EPA estimated future dredging for 14 years to reflect the 14 years of historic data provided by USACE.

Where:

PV = Present Value

Σ is the sum of the expression in brackets for the number of dredging occurrences of this dredging job

I = recurrence interval in years (including fractions of a year) for this dredging job

$14/I$ = number of periods between recurrence intervals for which costs will be discounted

Q_b = cubic yards dredged under the baseline scenario

R_{pc} = percentage of cubic yards remaining in the post-compliance scenario

C = cost per cubic yards dredged

d = annual discount rate[10]

$d*I$ = the discount rate for the period, being the annual rate d prorated for the length of the period I in years

n = the number of the specific dredging occurrence.

For example, if the Mississippi River is scheduled to be dredged every 2 years between 2010 and 2023, there will be no dredging costs for year 1, but the dredging costs in year 2 will be equal to the cost per cubic yard multiplied by the amount of sediment dredged, then discounted by 2 years. There will again be no dredging costs for year 3. Dredging costs will then be incurred in year 4 as in year 2, but discounted by 4 years, and so on. By the end of 2023, the Mississippi River will have been dredged seven times. The sum of the discounted costs of these six dredging occurrences is the total present value of the dredging cost of the Mississippi River for the next 14 years. Repeating this process for every dredging job in this analysis yields an estimate of national dredging costs over the next 14 years.

Alternately, the frequency can be interpreted as occurring at the beginning of each interval rather than at the end, so that in the example above, the Mississippi River would be dredged in year 1, then in year 3, etc., and discounted accordingly. This approach was taken for EPA's high estimate (see below).

7.4.3 Sensitivity Analysis

To account for uncertainty in projecting future dredging costs, EPA adjusted the assumptions described in *Section 7.2* to provide a range of benefits estimates. EPA estimated low, midpoint, and high levels of dredging activity for this sensitivity analysis as follows:

Low estimate:

- Assuming a minimum recurrence interval (below which occurrences are assumed to be continuations of the previous occurrence of the same job) of 180 days

10 EPA estimated costs using both 3 and 7 percent discount rates, in accordance with OMB (2003, 33-34).

- Assigning 10th percentile cost and cubic yards dredged estimates for jobs lacking these data
- Excluding jobs with only one occurrence over the 14 years from 1995 to 2008.

Midpoint estimate:

- Assuming a minimum recurrence interval of 90 days to include jobs with more closely spaced recurrences
- Using 50th percentile (median) cost and cubic yards dredged estimates for jobs lacking these data
- Including jobs with only one occurrence over the 14 years from 1995 to 2008 under the assumption that these jobs or similar jobs will occur once over the next 14 years.

High estimate:

- Assuming a minimum recurrence interval of 30 days
- Using 90th percentile cost and cubic yards dredged estimates for jobs lacking these data
- Including jobs with only one occurrence between 1995 and 2008 under the assumption that these jobs or similar jobs will occur once over the next 14 years.

This analysis also presents costs for all estimates using both 3 and 7 percent discount rates.

7.4.4 Annualizing Future Dredging Costs

Though dredging costs will not be incurred at a regular annual rate, EPA annualized its 14-year cost projection in order to facilitate the comparison of these costs to other costs calculated for the analysis of the regulation that are measured on an annual timeframe. This annualized cost was calculated using the same interest rates used to discount future dredging costs. *Table 7-5* presents estimated total annualized dredging (cubic yards removed) and the estimated costs for navigable waterway dredging for the baseline scenario, including low, midpoint, and high estimates.

Under the baseline scenario, midpoint dredging costs projected over the period of 2010 to 2023 are expected to range between $468 and 620 million per year (annualized using a 7 and a 3 percent discount rate, respectively). The high and low estimates produce an overall range between $360 million and $655 million. Regions 4 and 6 are expected to incur the highest costs, as they have the largest percentage of dredging activity and amounts of sediment dredged. In the low estimate, Region 8 incurs no dredging costs because the sole dredging job recorded in Region 8 in the USACE Dredging Information System occurs only once and was thus excluded. Regions 1 and 7 also have considerably lower dredging costs than other regions, due to the small number of recorded dredging jobs in these areas.

Table 7-5: Annualized Dredging Costs Under the Baseline Scenario (millions of 2008$)

EPA Region	Sediment Dredged (millions of cubic yards)			Cost using 3% discount rate (millions of 2008$)			Cost using 7% discount rate (millions of 2008$)		
	Low	Mid	High	Low	Mid	High	Low	Mid	High
1	1.4	3.7	6.6	$1.4	$3.0	$3.6	$1.0	$2.2	$2.6
2	84.5	111.5	130.5	$39.4	$54.1	$56.9	$29.8	$40.6	$42.6
3	95.5	124.6	134.8	$47.5	$61.6	$63.2	$36.2	$46.5	$47.6
4	405.5	540.8	575.9	$160.9	$213.2	$221.3	$122.4	$161.0	$167.4
5	85.4	112.7	149.2	$23.6	$34.9	$44.6	$17.9	$26.2	$33.5
6	1,240.5	1,483.3	1,503.8	$148.4	$185.6	$191.5	$112.3	$140.3	$145.4
7	7.4	17.3	18.8	$1.0	$2.5	$2.8	$0.7	$1.8	$2.0
8	0.0	0.5	0.5	$0.0	$0.1	$0.1	$0.0	$0.1	$0.1
9	60.5	67.7	82.3	$27.6	$31.1	$33.1	$21.1	$23.6	$25.2
10	112.2	153.5	178.3	$24.7	$34.2	$37.6	$18.9	$25.8	$28.4
Total	**2,093.0**	**2,615.7**	**2,780.8**	**$474.5**	**$620.4**	**$654.7**	**$360.3**	**$468.1**	**$494.9**

7.5 Estimating the Avoided Costs from Decreased Dredging of Navigable Waterways

The avoided costs for each post-compliance scenario are calculated as the difference in total annualized dredging costs between the baseline and each post-compliance scenario, and are considered the benefits to navigation resulting from the regulation. *Table 7-6, Table 7-7, Table 7-8,* and *Table 7-9* present annualized avoided costs from reduced dredging of navigable waterway for each of the Agency's policy options, including low, midpoint, and high estimates for cost reductions under each policy option. Each of these estimates was calculated using both 3 and 7 percent rates to discount and annualize costs. Because the discount rate does not make a significant difference in the overall avoided costs, all values discussed are those calculated assuming a 3 percent discount rate.

Annualized savings from reduced dredging activity range from $1.0 to $3.4 million. EPA estimates that Regions 4 and 6 will benefit from the most substantial reductions in dredging costs under all policy options. This is due to a large amount of dredging activity in these regions, and a large percentage reduction in sediment runoff expected as a result of the regulation. Due to the lack of significant dredging activity in Region 8, no noticeable benefits are expected in this region.

Option 1, which requires non-numeric effluent limitations for all sites, is EPA's least stringent policy option. It is predicted to produce a range of avoided costs between $1.0 and $1.3 million with a midpoint estimate of slightly less than $1.3. EPA predicts that this option would prevent 8.5 million cubic yards of sediment from entering navigable waters each year.

Option 2 requires ATS on sites with more than 30 acres disturbed at one time and imposes a 13 NTU turbidity standard while requiring non-numeric effluent limitations on all sites and is similar to the option EPA proposed previously in 2008. This option will prevent an estimated 17.6 million cubic yards of sediment from entering navigable water bodies and requiring dredging. The midpoint estimate for avoided costs under this option is $2.6 million per year, ranging from $2.1 to $2.8 million between the low and high estimates.

Option 3, EPA's most stringent policy option, requires ATS on sites with more than 10 acres disturbed at one time, imposes a 13 NTU turbidity standard on these sites, and requires non-numeric effluent limitations on all sites. This option would prevent approximately 22.0 million cubic yards of sediment from building up in navigable waterways each year. Avoided costs from this action range from $2.7 to $3.4 million, with a midpoint of $3.3 million.

Option 4 requires passive treatment systems on all sites with 10 or more acres disturbed at one time, and establishes a numeric turbidity standard of 250 NTU (expressed as a daily maximum value) for sites required to implement passive treatment. In addition, all sites will be required to meet non-numeric effluent limitations. Avoided costs from Option 4 range between $2.4 and $3.0 million, with a midpoint estimate of $2.9 million. The requirements of Option 4 produce larger reductions in dredged sediment than those of Option 2, as turbidity treatment is required on more sites. Option 4 is not as effective as Option 3 in reducing sediment in navigable waters, since the two options have the same criteria for disturbed acres, but Option 4 has a less stringent turbidity standard. The total reduction in sediment dredged from navigable waters is expected to be 20.0 million cubic yards under Option 4.

Table 7-6: Reductions in Dredging and Annualized Avoided Costs Under Option 1

EPA Region	Reduction in Sediment Dredged (thousands of cubic yards)			Avoided Costs Using 3% Discount Rate (thousands of 2008$)			Avoided Costs Using 7% Discount Rate (thousands of 2008$)		
	Low	Mid	High	Low	Mid	High	Low	Mid	High
1[1]	0.3	1.0	1.3	$0.3	$0.8	$0.9	$0.3	$0.8	$0.9
2	16.4	21.3	24.2	$7.6	$10.4	$10.9	$7.4	$10.1	$10.5
3	40.7	51.5	55.0	$20.0	$25.0	$25.6	$19.7	$24.4	$24.9
4	871.5	1,001.7	1,058.3	$262.5	$320.0	$329.8	$258.1	$312.5	$322.3
5	21.0	31.5	50.9	$8.3	$12.7	$18.3	$8.2	$12.4	$17.9
6	5,901.7	6,660.6	6,739.7	$628.8	$739.1	$769.7	$611.5	$720.8	$754.8
7	5.2	9.3	10.6	$0.8	$1.6	$1.7	$0.7	$1.5	$1.6
8	0.0	0.0	0.0	$0.0	$0.0	$0.0	$0.0	$0.0	$0.0
9	56.4	81.6	95.5	$28.7	$41.3	$43.3	$28.2	$40.0	$41.9
10	486.8	614.8	795.1	$69.7	$106.2	$131.8	$68.3	$102.9	$127.5
Total	**7,400.0**	**8,473.3**	**8,830.6**	**$1,026.7**	**$1,257.2**	**$1,331.9**	**$1,002.4**	**$1,225.3**	**$1,302.4**

Table 7-7: Reductions in Dredging and Annualized Avoided Costs Under Option 2

EPA Region	Reduction in Sediment Dredged (thousands of cubic yards)			Avoided Costs Using 3% Discount Rate (thousands of 2008$)			Avoided Costs Using 7% Discount Rate (thousands of 2008$)		
	Low	Mid	High	Low	Mid	High	Low	Mid	High
1	0.7	2.1	2.7	$0.7	$1.7	$1.9	$0.7	$1.6	$1.8
2	33.8	43.8	49.9	$15.6	$21.4	$22.4	$15.2	$20.7	$21.7
3	84.5	106.7	114.0	$41.4	$51.9	$53.0	$40.8	$50.7	$51.7
4	1,806.7	2,076.3	2,193.3	$543.1	$661.9	$682.2	$534.0	$646.4	$666.7
5	43.2	65.0	105.1	$17.1	$26.2	$37.6	$16.8	$25.5	$36.8
6	12,309.2	13,891.9	14,056.6	$1,311.4	$1,541.4	$1,605.1	$1,275.4	$1,503.1	$1,574.1
7	10.8	19.4	22.0	$1.6	$3.2	$3.6	$1.5	$3.1	$3.4
8	0.0	0.1	0.1	$0.0	$0.0	$0.0	$0.0	$0.0	$0.0
9	116.9	168.5	197.4	$59.5	$85.4	$89.4	$58.5	$82.6	$86.6
10	990.2	1,251.4	1,618.7	$141.8	$216.1	$268.1	$138.8	$209.4	$259.4
Total	**15,395.9**	**17,625.0**	**18,359.8**	**$2,132.1**	**$2,609.2**	**$2,763.4**	**$2,081.7**	**$2,543.0**	**$2,702.2**

Table 7-8: Reductions in Dredging and Annualized Avoided Costs Under Option 3

EPA Region	Reduction in Sediment Dredged (thousands of cubic yards)			Avoided Costs Using 3% Discount Rate (thousands of 2008$)			Avoided Costs Using 7% Discount Rate (thousands of 2008$)		
	Low	Mid	High	Low	Mid	High	Low	Mid	High
1	0.9	2.6	3.4	$0.8	$2.1	$2.3	$0.8	$2.0	$2.2
2	42.1	54.5	62.1	$19.4	$26.7	$27.9	$18.9	$25.8	$27.0
3	105.3	132.9	142.0	$51.6	$64.7	$66.0	$50.8	$63.1	$64.5
4	2,251.4	2,587.4	2,733.0	$676.6	$824.5	$849.8	$665.2	$805.1	$830.4
5	53.8	81.0	130.8	$21.2	$32.6	$46.9	$20.9	$31.8	$45.7
6	15,356.2	17,330.6	17,536.2	$1,636.0	$1,922.9	$2,002.4	$1,591.0	$1,875.1	$1,963.7
7	13.5	24.1	27.5	$2.0	$4.0	$4.5	$1.9	$3.8	$4.3
8	0.0	0.1	0.1	$0.0	$0.0	$0.0	$0.0	$0.0	$0.0
9	145.7	209.8	245.9	$74.2	$106.3	$111.3	$72.9	$102.9	$107.8
10	1,229.5	1,554.0	2,010.3	$176.0	$268.4	$333.0	$172.4	$260.0	$322.1
Total	**19,198.2**	**21,977.0**	**22,891.2**	**$2,657.7**	**$3,252.1**	**$3,444.1**	**$2,594.9**	**$3,169.7**	**$3,367.8**

Table 7-9: Reductions in Dredging and Annualized Avoided Costs Under Option 4

EPA Region	Reduction in Sediment Dredged (thousands of cubic yards)			Avoided Costs Using 3% Discount Rate (thousands of 2008$)			Avoided Costs Using 7% Discount Rate (thousands of 2008$)		
	Low	Mid	High	Low	Mid	High	Low	Mid	High
1	0.6	1.7	2.3	$0.6	$1.4	$1.6	$0.6	$1.3	$1.5
2	33.7	43.5	49.6	$15.5	$21.3	$22.3	$15.1	$20.6	$21.6
3	92.1	115.9	123.8	$45.2	$56.5	$57.7	$44.5	$55.2	$56.3
4	1,964.1	2,252.6	2,374.9	$574.7	$699.0	$720.3	$565.1	$682.8	$704.1
5	43.4	67.2	107.2	$16.9	$26.1	$37.2	$16.7	$25.5	$36.3
6	14,395.8	16,242.8	16,434.8	$1,531.7	$1,800.0	$1,874.4	$1,489.6	$1,755.3	$1,838.3
7	12.4	22.3	25.3	$1.8	$3.7	$4.2	$1.8	$3.5	$3.9
8	0.0	0.1	0.1	$0.0	$0.0	$0.0	$0.0	$0.0	$0.0
9	127.8	174.1	205.7	$65.1	$88.3	$92.6	$64.0	$85.6	$89.9
10	815.4	1,043.3	1,351.7	$116.0	$178.8	$222.5	$113.6	$173.1	$215.1
Total	**17,485.3**	**19,963.5**	**20,675.5**	**$2,367.5**	**$2,875.1**	**$3,032.8**	**$2,310.9**	**$2,802.9**	**$2,967.1**

7.6 Sources of Uncertainty and Limitations

Key limitations of EPA's analysis of cost savings to drinking water treatment facilities are outlined below. The SPARROW model for suspended sediments that was used for estimating TSS concentrations in the source waters also has a number of limitations. These limitations are discussed in detail in *Chapter 6.*

- The USACE dredging database identifies dredging jobs by name, which is usually the name of the waterbody dredged. However, the data lack standardized naming conventions, so it is possible that the same waterbody is dredged under different job names. This may result in the exclusion of dredging job names that only appear once in the database, but in fact were carried out in the same water bodies as a differently named job, which would result in a downward bias in EPA's dredging frequency calculations and the project costs.
- The navigable waterway data provide latitude/longitude information for some dredging jobs, which are used to link dredging jobs to RF1 reaches, but these data are incomplete. In cases where latitude/longitude information was not available for a particular job, EPA matched it to an RF1 reach using the job name. This is a potential source of inaccuracy, as the job name is often the waterway name, and may not be very specific (in cases such as the Mississippi or Colorado rivers). Additionally, if no RF1 reach could be identified for the dredging job, the average reduction for the state in which the job occurred was used. It is unclear whether this would lead to an over- or underestimate of benefits.

8 Benefits to Water Storage

Reservoirs are water impoundments, often manmade, that serve many functions, including providing drinking water, flood control, hydropower supply, and recreational opportunities. Sediment in streams can be carried into reservoirs, where it may settle and build up layers of silt over time. An increase in sedimentation rates will reduce the useful life of a reservoir, unless measures are taken to reclaim some of its capacity. Historically, the United States Geological Survey (USGS) has recorded an average of 1.2 billion kilograms of sediment deposition per reservoir each year (USGS 2007c).

The regulation is expected to reduce the amount of sediment entering reservoirs and, as a result, the cost of reservoir maintenance. Though there are multiple options for sediment mitigation, based on a literature review EPA used the avoided cost of reservoir dredging as a measure of the benefits of the regulation to water storage. This approach represents an appropriate measure of social benefits for cases in which a policy change reduces costs to producers (in this case producers of water supplies for human use), but in which price effects are minimal (cf. Boardman et al. 2001, p. 70-74). A description of other amelioration measures can be found in *Section 8.1*.

The analysis of these benefits includes the following steps:

- Estimating the unit cost of sediment removal from reservoirs
- Estimating the sediment accumulation in reservoirs under the baseline scenario and the post-compliance EPA policy options
- Estimating the cost that would be incurred to dredge this accumulated sediment under each scenario and the avoided cost of reservoir dredging expected from this regulation.

The remainder of this chapter provides details of this analysis.

8.1 Review of Literature on Reservoir Sedimentation

Research into reservoir sedimentation was conducted in the 1980s as part of a larger research effort into the economic effects of sediment runoff from agricultural land use. As sediment has similar effects regardless of its source, many of the insights from these studies will be applicable to sediment discharge from construction activities.

Crowder (1987) estimated that the United States was losing about 0.22 percent of its reservoir capacity each year due to sedimentation. Clark et al. (1985) notes that total U.S. reservoir capacity is filling up slowly and has enough excess capacity dedicated to hold sediment build-up over hundreds of years. However, the study goes on to conclude that while total reservoir sedimentation is manageable, sedimentation is far from uniform and that in about 15 percent of U.S. reservoirs, sedimentation rates exceeded 3 percent of capacity annually, and in the more extreme cases, 10 percent per year (Clark et al. 1985). *The Reservoir Sedimentation Handbook* (Morris and Fan 1997), an engineering guide, details several methods of counteracting the effects of sedimentation to maintain reservoir functionality:

- *Sediment routing* encompasses a group of techniques that seek to allow sediment-laden water to pass around or through the reservoir, not allowing the sediment to settle. All of these techniques involve timing sediment-laden flows from events such as storms and floods and operating the pass-through or bypass mechanism optimally. Pass-through techniques may be implemented at existing reservoirs; however, the bypass mechanisms are most cost-effective if incorporated into the initial design of the reservoir. SPARROW does not take into account any of these techniques in its assumptions about sediment removed from reaches by reservoirs, EPA was not able to find any information on either the prevalence or effectiveness of these techniques.

- *Flushing* involves scouring accumulated sediment from the reservoir by partially or fully draining the reservoir and allowing the erosive force of the water to carry the sediment through the dam and downstream of the reservoir. This does not appear to be a very common practice in the United States, as this creates extraordinarily high sediment concentrations downstream of the flushed reservoir, sometimes in excess of 1,000,000 mg/L and thus may require special environmental permission according to Morris and Fan (1997). Flushing is not one-hundred percent effective at removing settled sediment, and the reservoir may still require dredging. The process also requires interrupting the service of the reservoir.
- *Dredging* is a practical and common approach to sediment removal once a reservoir has been built. Morris and Fan (1997) note that with the exception of draining a reservoir and excavating settled sediment (which is often more expensive and less common than dredging), dredging may be the only feasible option for sediment removal in many reservoirs. Dredging may be limited by available funds, as well as by the amount of space available to dispose of the dredged material. Due to the cost of transporting dredged sediment, a lack of nearby disposal sites may limit the amount of sediment that can be removed using dredging.

In a study focused on the economic costs of reservoir sedimentation due to agricultural runoff, Crowder (1987) describes several options for mitigating reservoir sedimentation. These options are:

- Including sediment pools in the initial construction of the reservoir to collect sediment entering the reservoir and maintaining these pools to increase the useful life of the reservoir
- Dredging the reservoir
- Replacing the lost capacity by expanding or replacing the reservoir.

According to Crowder (1987), constructing sediment pools at the outset is more than five times less costly than dredging the reservoir. This study estimates that reservoir capacity costs between $564 and $1,316 (2008$) per acre-foot to construct (regardless of whether it is built as excess capacity for sediment or to replace capacity lost from sedimentation), and that dredging costs $4,700 (2008$) per acre-foot of material dredged. He estimates that only 20,000 (32.3 million cubic yards) acre-feet of sediment are removed from reservoirs each year by dredging, while sediment pools in reservoirs accumulate 1 million acre-feet (1.6 billion cubic yards) of sediment per year, and that new capacity would have to account for the remaining 620,000 acre-feet (1 billion cubic yards) lost to sediment each year (Crowder 1987). His estimate of total economic damages from reservoir sedimentation is $1.3 billion (2008$) per year, based on an assumed $94 million (2008$) in annual dredging expenditures and the discounted cost of replacing all remaining lost capacity in 10 to 20 years.

The analysis used by Crowder (1987) assumes that all reservoir sediment that is not dredged or accumulated by sediment pools is accounted for with the construction of new capacity. This assumption implies that additional reservoir capacity can be built unlimitedly to maintain the useful capacity of reservoirs, an assumption that the author acknowledges is unrealistic. Although Crowder (1987) finds that building excess storage capacity may be more cost effective than dredging reservoirs, it may not be a feasible option in many cases. Moreover, in comparing costs of reservoir dredging and building new reservoir capacity, Crowder (1987) does not take into account ecological effects of new reservoir capacity construction. The value of ecological services lost due to new reservoir construction may outweigh avoided costs compared to reservoir dredging. The study does also not take into account any opportunity costs of increasing the amount of land used for water storage. The objective of Crowder (1987) is to quantify the impact of sedimentation on water storage reservoirs by assuming that all capacity lost to sedimentation that is not currently recovered through dredging will need to be replaced by new capacity. While this approach is appropriate for a theoretical estimation of the cost impact of reservoir sedimentation, EPA's objective is to quantify the avoided costs of *reduced* reservoir sedimentation due to reduced construction site stormwater discharges, and thus only takes into account actual measures being taken to reduce reservoir sedimentation (i.e., dredging). Therefore, this analysis assumes that excess sediment is removed from reservoirs by dredging rather than building new reservoir capacity.

8.2 Data Sources

Information on reservoir locations, uses, and historic sedimentation rates is available from the Reservoir Sedimentation Survey Information System (RESIS) database (USGS 2007b). Formally known as RESIS-II, this database was initially developed by the U.S. Department of Agriculture Natural Resources Conservation Service, and was provided to EPA by USGS.

The database includes 4,227 sedimentation survey results from 1,819 reservoirs located across the coterminous United States. Results from 1775 to 1992 are represented, with the majority of measurements covering the mid-20th century and on. RESIS does not include all U.S. reservoirs, nor does it contain information about dredging activity in reservoirs or about any other methods for recapturing lost water storage capacity.

Table 8-1 details sediment deposition per year recorded by RESIS in each EPA region. More than 1.2 billion kilograms of sediment are deposited in the average U.S. reservoir each year. The most sediment deposition is recorded in Region 6 with 6.6 billion kilograms per reservoir per year, almost four times more than the national average. Region 9 also reports large masses of sediment deposition each year of 1.8 billion kilograms per reservoir, which is still greater than the national average. Other regions have average annual deposition per reservoir ranging from 800,000 kilograms in Region 1 to 440 million in Region 4.

Table 8-1: Average Annual Sedimentation Deposition Recorded by RESIS

EPA Region	Number of Reservoirs	Average Annual Sediment Deposition per Reservoir (million kg/year)
1	9	0.8
2	18	25.1
3	81	71.5
4	168	440.4
5	272	25.9
6	338	6,568.9
7	329	57.2
8	208	436.0
9	285	1,817.6
10	110	67.2
Total	**1,818**	**1,234.0**

Source: RESIS (USGS 2007b).

Changes in sediment attenuation in reservoirs will be predicted by the SPARROW model, the details of which can be found in *Chapter 6*.

8.3 Estimating the Unit Cost of Sediment Removal from Reservoirs

Because RESIS does not provide information on reservoir dredging, EPA assumed for the purpose of this analysis that the average unit cost of dredging sediment from a navigable waterway is approximately equal to the average unit cost of dredging sediment from a reservoir. Crowder (1987) does provide an estimate of reservoir dredging cost per unit of sediment dredged, but gives no empirical basis for this estimate. Because the USACE dredging database provides costs and quantity dredged for more than 3,300 dredging jobs in the past 12 years, this unit cost of dredging has more robust empirical foundations. EPA does not expect there to be any significant difference between the costs of reservoir dredging and the costs of navigable waterway dredging, as all of the variable costs for dredging are related to the quantity of sediment dredged. Morris and Fan (1997) cite a unit cost of $4.45 (2008$) per cubic yard for sediment dredging at Lake Springfield, Illinois, which is quite close to the $4.41 average cost per cubic yard for EPA Region 5 in the navigational dredging data. *Section 7.3* provides details on estimating the unit cost of navigable waterway dredging that will be used as a surrogate for the unit cost of reservoir dredging. This analysis will use average costs at the regional level.

8.4 Estimating the Total Cost of Reservoir Dredging Under Different Policy Options

8.4.1 Estimating Sediment Accumulation in Reservoirs and the Amount of Sediment Expected To Be Dredged

EPA estimated the annual sediment attenuation rate in each reservoir under the baseline and the three policy options using output data from the SPARROW model. *Chapter 6* provides a description of the model. This analysis uses data output from SPARROW that predicts the amount of sediment (in kilograms) settling in reservoirs in a reach. To estimate the amount of sediment flowing into and settling in reservoirs in each region, EPA summed the values in this field under the baseline scenario and each policy option.[11] The amount of sediment settling in reservoirs under each scenario forms the basis for the estimation of the quantity of sediment dredged from reservoirs and the costs of this dredging.

For the purposes of this analysis, EPA assumed, consistent with Crowder (1987), that all sediment entering reservoirs must be removed in order to maintain the current water storage capacity in the United States. The frequency of dredging is highly site-specific, depending on many factors including the average sediment concentration of the river or stream, the size of the reservoir and excess storage capacity, and any sediment routing practices. For this analysis, EPA chose a general frequency of reservoir dredging based on information presented by the USACE in a Final Dredged Material Management Plan and Environmental Impact Statement for reservoirs in Washington (USACE 2002). This report states that “dredging cycles may vary from 2 to 10 years” (USACE 2002, p. 66). EPA used these frequencies as high and low estimates, and employed their arithmetic mean of 6 years for a midpoint

[11] EPA conducted a preliminary analysis of baseline costs including and excluding outliers, and found that the exclusion of outliers does not have a significant effect on the total cost of reservoir dredging. The values used for this analysis include reservoir sedimentation for all reservoirs on reaches modeled by SPARROW.

estimate. This approach provides a range of benefits estimates to account for uncertainty in the frequency of reservoir dredging.

8.4.2 Estimating the Total Cost of Reservoir Dredging Under the Baseline Scenario and Policy Options

The cost of dredging the sediment settling out in reservoirs is estimated by using the average dredging cost per region as described in *Section 8.3*. Because this cost is given per cubic yard, the sediment attenuation in reservoirs given by SPARROW will be converted from kilograms to cubic yards using a sediment density of 1.5 g/cm^3 (Hargrove 2007). This translates to a conversion of 1,147 kilograms per cubic yard. The equation below summarizes the calculation of costs for one cycle of dredging before discounting and annualization.

$$TC_{reservoirs} = \sum_{R=1}^{10} \left[[I] * \left(\frac{[S_R]}{1{,}147} \right) * [C_R] \right] \qquad \text{(Eq. 8-1)}$$

Where:

$TC_{reservoirs}$ = total cost of dredging all sediment settling in reservoirs in all regions

R = region number

I = the assumed interval in years of reservoir dredging; varied between 2, 6, and 10

S_R = sum of all reservoir sediment settlement predicted by SPARROW in a region in kilograms

1,147 = kilograms in a cubic yard

C_R = regional average of historic dredging job cost per cubic yard, 1995-2006

The resulting costs were discounted and annualized over the assumed interval using both 3 and 7 percent discount rates, in accordance with OMB Circular A-4 (OMB 2003).

Table 8-2 presents the total amount of sediment that is estimated to be dredged per year and the estimated cost of this dredging under the baseline scenario, including low, midpoint, and high estimates. The 445 million cubic yards predicted by SPARROW to settle annually in reservoirs is expected to cost between \$1.5 and \$1.7 billion to dredge under the baseline scenario. Region 8 has the highest dredging costs, and though less sediment is predicted to accumulate in Region 8 reservoirs than in those in Region 6, the former has a much higher unit cost of dredging. Regions 1, 2, and 3 are all predicted to accumulate less than 6 million cubic yards of sediment per year each, and accordingly are estimated to have lower overall dredging costs. For other regions, midpoint estimates of dredging costs range between \$50 and \$526 million.

Table 8-2: Estimated Cost of Reservoir Dredging Under the Baseline Scenario

EPA Region	Sediment Dredged (yd^3)	Estimated Cost (millions of 2008$)		
		Low	Mid	High
1	1,278,280	$11.7	$12.4	$13.2
2	2,725,285	$17.1	$18.2	$19.3
3	5,337,766	$32.3	$34.4	$36.5
4	45,666,958	$218.0	$231.8	$246.2
5	35,940,396	$147.1	$156.4	$166.2
6	128,381,390	$201.3	$214.0	$227.3
7	42,237,651	$96.0	$102.1	$108.4
8	126,434,846	$495.0	$526.4	$559.1
9	41,859,743	$236.2	$251.2	$266.8
10	14,743,876	$46.9	$49.9	$53.0
Total	**444,606,190**	**$1,501.6**	**$1,596.8**	**$1,696.0**

8.5 Estimating the Avoided costs from Reduced Reservoir Dredging

The difference between the anticipated dredging costs under the baseline and a particular policy option represents the avoided costs of that policy option. *Table 8-3, Table 8-4, Table 8-5*, and *Table 8-6* present reductions in sedimentation and subsequent avoided costs from reduced reservoir dredging for each policy option, including low, midpoint, and high estimates under these options. Because the range of estimates is relatively small between the low and high estimates, the values presented below are midpoint estimates unless otherwise stated. Benefits were estimated assuming both 3 and 7 percent discount rates, consistent with OMB Circular A-4 (OMB 2003).

Avoided costs from a reduction in reservoir sedimentation range from $1,072 (Option 1, low estimate, 7 percent discount rate) to $3.6 million (Option 3, high estimate, 3 percent discount rate), depending on the policy option, the assumed frequency of reservoir dredging, and the discount rate. The largest savings are predicted in Region 6 under all options, as SPARROW predicts the largest overall reductions in sediment accumulation in this region. Region 4 is also expected to benefit substantially relative to other regions due to both large reductions in construction discharges in this region and a relatively high unit cost of dredging (estimated from USACE data).

Option 1, which requires non-numeric effluent limitations for all sites, is EPA's least stringent policy option. This option is expected to reduce reservoir sedimentation by about 560 thousand cubic yards nationally every year, which is estimated to save between $1.2 (7 percent discount rate) and $1.4 million (3 percent discount rate) in dredging costs per year.

Option 2 requires ATS on sites with more than 30 acres disturbed at one time and imposes a 13 NTU turbidity standard while requiring non-numeric effluent limitations on all sites and is similar to the option EPA proposed previously. This action is estimated to prevent about 1.2 million cubic yards of sediment from building up in reservoirs each year saving between $2.6 (at a 7 percent discount rate) and $2.9 (at a 3 percent discount rate) million in dredging costs per year.

Option 3, EPA's most stringent policy option, requires ATS on sites with more than 10 acres disturbed at one time, imposes a 13 NTU turbidity standard on these sites, and requires non-numeric effluent limitations on all sites. Its reduction in sediment deposition in reservoirs is estimated to be 1.5 million cubic yards annually. The estimated avoided costs of reduced dredging are between $3.2 and $3.6 million.

Option 4 requires passive treatment systems on all sites with 10 or more acres disturbed at one time, and establishes a numeric turbidity standard of 250 NTU (expressed as a daily maximum value) for sites required to implement passive treatment. In addition, all sites will be required to meet non-numeric effluent limitations. Because Option 4 requires passive rather than active treatment of sediment and has a higher turbidity standard, the estimated reductions in reservoir sediment buildup are lower than those from Option 3. However, the estimated reductions from Option 4 are still greater than those expected from Option 2, as that option requires treatment on fewer sites. Expected avoided costs from Option 4 are estimated to be $2.9 million assuming a 7 percent discount rate, and $3.2 million assuming a 3 percent rate. Regions 4 and 6 together account for more than half of the monetized benefits of reduced reservoir sedimentation, due to the expected reductions in construction discharges in these regions. Option 4 is estimated to reduce reservoir sedimentation by 1.3 million cubic yards annually.

Table 8-3: Reduction in Reservoir Dredging and Avoided Costs Under Option 1

EPA Region	Annual Reduction in Sediment yd^3)	Avoided Costs (thousands of 2008$)					
		3% Discount Rate			7% Discount Rate		
		Low	Mid	High	Low	Mid	High
1	1,423	$13.0	$13.8	$14.7	$10.8	$12.5	$14.4
2	535	$3.4	$3.6	$3.8	$2.8	$3.2	$3.7
3	2,554	$15.5	$16.5	$17.5	$12.8	$14.9	$17.1
4	98,832	$471.7	$501.6	$532.8	$391.4	$453.6	$522.5
5	7,878	$32.2	$34.3	$36.4	$26.8	$31.0	$35.7
6	408,507	$640.5	$681.1	$723.4	$531.4	$615.9	$709.4
7	24,768	$56.3	$59.8	$63.6	$46.7	$54.1	$62.3
8	2,165	$8.5	$9.0	$9.6	$7.0	$8.2	$9.4
9	4,833	$27.3	$29.0	$30.8	$22.6	$26.2	$30.2
10	7,293	$23.2	$24.7	$26.2	$19.3	$22.3	$25.7
Total	**558,788**	**$1,291.5**	**$1,373.4**	**$1,458.7**	**$1,071.6**	**$1,241.9**	**$1,430.5**

Table 8-4: Reduction in Reservoir Dredging and Avoided Costs Under Option 2

EPA Region	Annual Reduction in Sediment (yd^3)	Avoided Costs (thousand 2008$)					
		3% Discount Rate			7% Discount Rate		
		Low	Mid	High	Low	Mid	High
1	2,876	$26.3	$28.0	$29.7	$21.8	$25.3	$29.1
2	1,096	$6.9	$7.3	$7.8	$5.7	$6.6	$7.6
3	5,278	$32.0	$34.0	$36.1	$26.5	$30.7	$35.4
4	205,148	$979.1	$1,041.2	$1,105.8	$812.4	$941.5	$1,084.5
5	16,292	$66.7	$70.9	$75.3	$55.3	$64.1	$73.9
6	851,211	$1,334.6	$1,419.2	$1,507.4	$1,107.4	$1,283.3	$1,478.2
7	51,568	$117.2	$124.6	$132.3	$97.2	$112.7	$129.8
8	4,476	$17.5	$18.6	$19.8	$14.5	$16.8	$19.4
9	9,992	$56.4	$60.0	$63.7	$46.8	$54.2	$62.5
10	14,804	$47.1	$50.1	$53.2	$39.1	$45.3	$52.2
Total	**1,162,741**	**$2,683.8**	**$2,853.8**	**$3,031.2**	**$2,226.8**	**$2,580.6**	**$2,972.6**

Table 8-5: Reduction in Reservoir Dredging and Avoided Costs Under Option 3

EPA Region	Annual Reduction in Sediment (yd^3)	Avoided Costs (thousand 2008$)					
		3% Discount Rate			7% Discount Rate		
		Low	Mid	High	Low	Mid	High
1	3,567	$32.6	$34.7	$36.9	$27.1	$31.4	$36.1
2	1,363	$8.5	$9.1	$9.7	$7.1	$8.2	$9.5
3	6,573	$39.8	$42.3	$45.0	$33.0	$38.3	$44.1
4	255,706	$1,220.4	$1,297.7	$1,378.4	$1,012.6	$1,173.5	$1,351.7
5	20,293	$83.1	$88.3	$93.8	$68.9	$79.9	$92.0
6	1,061,735	$1,664.7	$1,770.2	$1,880.2	$1,381.2	$1,600.7	$1,843.8
7	64,312	$146.1	$155.4	$165.0	$121.2	$140.5	$161.9
8	5,575	$21.8	$23.2	$24.7	$18.1	$21.0	$24.2
9	12,446	$70.2	$74.7	$79.3	$58.3	$67.5	$77.8
10	18,375	$58.5	$62.2	$66.1	$48.5	$56.2	$64.8
Total	**1,449,945**	**$3,345.8**	**$3,557.9**	**$3,778.9**	**$2,776.1**	**$3,217.2**	**$3,705.9**

Table 8-6: Reduction in Reservoir Dredging and Avoided Costs Under Option 4

EPA Region	Annual Reduction in Sediment (yd^3)	Avoided Costs (thousand 2008$)					
		3% Discount Rate			7% Discount Rate		
		Low	Mid	High	Low	Mid	High
1	2,126	$19.4	$20.7	$22.0	$16.1	$18.7	$21.5
2	1,022	$6.4	$6.8	$7.2	$5.3	$6.2	$7.1
3	5,497	$33.3	$35.4	$37.6	$27.6	$32.0	$36.9
4	226,512	$1,081.1	$1,149.6	$1,221.0	$897.0	$1,039.5	$1,197.4
5	17,172	$70.3	$74.7	$79.4	$58.3	$67.6	$77.9
6	984,437	$1,543.5	$1,641.3	$1,743.3	$1,280.7	$1,484.2	$1,709.6
7	59,072	$134.2	$142.7	$151.6	$111.4	$129.1	$148.7
8	4,697	$18.4	$19.6	$20.8	$15.3	$17.7	$20.4
9	10,505	$59.3	$63.0	$67.0	$49.2	$57.0	$65.7
10	11,828	$37.6	$40.0	$42.5	$31.2	$36.2	$41.7
Total	**1,322,867**	**$3,003.6**	**$3,193.9**	**$3,392.4**	**$2,492.1**	**$2,888.1**	**$3,326.8**

8.6 Sources of Uncertainty and Limitations

Key limitations of EPA's analysis of avoided costs to drinking water treatment facilities are outlined below. The SPARROW model for suspended sediments that was used for estimating TSS concentrations in the source waters also has a number of limitations. These limitations are discussed in detail in *Chapter 6.*

- The reliance of the SPARROW model on the RF1 network is a significant limitation because the reservoirs located off the RF1 network are omitted from this analysis. This omission is likely to bias the estimated savings from reduced reservoir dredging downward.
- There is uncertainty as to the uniformity of sediment density because it is related to the type of soil in the area. Using a single density to convert volume to weight for all sediment may reduce the accuracy of the resulting cost estimates. However, the direction of this potential bias is uncertain.

- The lack of data on reservoir dredging results in significant uncertainty as to the types of reservoirs that are dredged and the cost of this dredging. The appropriateness of the avoided cost approach is conditional upon the estimates representing actual dredging costs that are no longer imposed on reservoirs. If typical reservoirs do not reduce dredging, and hence costs, but rather undertake other actions in response to reduced sediment loads, then actual benefits may diverge from estimates provided above.
- Neither RESIS nor the SPARROW model take into account sediment build-up in natural water storage facilities such as glacial lakes and ponds, so any activity to mitigate sedimentation of these areas is not included in this analysis, resulting in a potential downward bias of this estimate.

9 Benefits to Drinking Water Treatment

Drinking water treatment plants provide the essential services of removing contaminants from water taken from surface and ground sources, making the water potable. Sediment is one of the contaminants that must be removed from drinking water before it can be consumed, so a reduction in sediment levels in water taken up by drinking water treatment plants is expected to result in lower treatment costs for these plants. This section describes the methodology EPA used to estimate the reduction in drinking water treatment costs that should result from a reduction of sediment discharge from construction sites. This approach represents an appropriate measure of social benefits for cases in which a policy change reduces costs to producers (in this case producers of drinking water), but in which price effects are minimal (cf. Boardman et al. 2001, p. 70-74).

9.1 Construction Discharge Effects on Drinking Water Treatment Costs

9.1.1 Sediment

The primary concern of sediment in terms of drinking water treatment is the turbidity that results from suspended sediment in the water column. As turbidity increases in drinking water influent, the variable costs of water treatment to mitigate this turbidity increase.

The variable costs associated with treating drinking water for turbidity are twofold:

- Adding chemicals coagulants that bond to the sediment particles and help them sink out of the water column, and
- Disposing of the sludge which results from this treatment.

Several attempts have been made in the past two decades to isolate these elements of drinking water treatment costs:

- Forster et al. (1987) estimated $172 (2008$) per million gallons for drinking water sediment treatment, while Holmes (1988) performed several estimates, with ranges between $7.45 and $212 (2008$) per million gallons. It is unclear in these studies what elements were included in these estimates.
- Moore and McCarl (1987) estimated a total of $41 (2008$) per million gallons to treat water and dispose of the sludge for drinking water treatment plants in Oregon.
- Dearmont et al. (1998) put the chemical costs of turbidity treatment alone at $96 (2008$) per million gallons for Texas drinking water.

This broad range of cost estimates for sediment and turbidity treatment may be due to the fact that the studies measure costs per million gallons of water, without respect to the specific sediment levels in drinking water influent.

9.1.2 Nutrients

In addition to removing sediment from drinking water, treatment plants must also meet standards for certain nutrients. Nitrogen and phosphorus are the two primary nutrients of concern in construction discharges. While there is no EPA standard for phosphorus levels in drinking water, nitrogen is regulated by EPA with maximum contaminant levels (MCLs) of 10 mg/L for nitrate and 1 mg/L for nitrite; total nitrogen (nitrate + nitrite) is limited to 10 mg/L (USEPA undated). DeZuane (1997) cites EPA surface water surveys that found that only 1.2 percent of surveyed supplies had total nitrogen concentrations exceeding 10 mg/L.

EPA recommends ion exchange, reverse osmosis, and electrodialysis to remove nitrogen from source water (USEPA undated). Reverse osmosis and electrodialysis are systems with operating costs related to the volume of water treated rather than the amount of contaminants, while ion exchange does have the variable cost of the ion exchange resin to which the contaminant bonds. Therefore, drinking water treatment facilities that use reverse osmosis and electrodialysis incur no additional cost associated with nitrogen removal. Although facilities that use ion exchange may incur operating costs associated with nitrogen removal, the process is also used to remove many other contaminants, and estimating the fraction of variable cost attributable to nitrogen removal is not feasible within the national-level analysis.

Benefits from reduced nitrogen loadings to drinking water source reaches would only be realized if their nitrogen loadings are above the MCL for nitrate and nitrite. *Table 9-1* presents total nitrogen (TN) concentrations as predicted by SPARROW for the reaches in the drinking water treatment benefits analysis. As shown, the top 4 percent of these reaches are predicted to have concentrations above the 10 mg/L threshold. However, because EPA's methodology treats values above the 95th percentile as outliers, benefits for these reaches would not be able to be monetized.

Table 9-1: TN Concentrations Predicted by SPARROW for Reaches Serving as Drinking Water Sources[1]

Distribution of TN Concentrations	TN Concentration (mg/L)
Minimum	0.0
25th percentile	0.9
50th percentile	1.3
75th percentile	2.4
95th percentile	8.0
96th percentile	*10.0*
Maximum	2,503.7
Average	4.8

[1] Only includes reaches that are both modeled by SPARROW and included in the federal drinking water database SDWIS, see *Section 9.5* for details.

9.1.3 Increase in Primary Productivity and Associated Water Quality Impacts from Nutrient Discharges (Eutrophication)

In addition to the meeting the nitrogen standards discussed above, indirect construction-related nutrient impacts can result in the development of harmful algal blooms (HABs) in surface drinking water reservoirs. These algal blooms, particularly those comprised of blue-green algae (Cyanobacteria), can pose potential health threats and aesthetic problems for the reservoir water's potability. These include the production of taste and odor compounds, production of cyanotoxins, and the potential for increase disinfection by-product formation.

Cyanobacteria and actinomycetes produce the organic compound *geosmin*, which is typically released on the death or lysis of the cell. This compound is responsible for the muddy smell and unpleasant taste in drinking water that originates from surface water bodies experiencing algal blooms. The human sensory system is highly sensitive to geosmin. Studies show that the nose can detect this compound at concentrations as low as 5 parts per trillion and that taste can detect geosmin at 0.7 parts per billion. The second taste and odor compound found in drinking water is 2-methylisoborneol (or MIB). It is also produced by blue-green algae and detected by humans at very low levels of concentration. Together, these two compounds are responsible for the taste and odor associated with drinking water treatment, which can be quite strong in the presence of algal bloom. Taste and odor problems associated with drinking water are a pervasive and significant problem for many municipalities. In 1996, a survey of 377 American and Canadian water utilities conducted by the American Water Works Association reports that 43 percent of the utilities had a taste and odor problem lasting more than one week and 16 percent consider their drinking water taste and odor problem to be serious (Suffet et al. 1996).

Large concentrations of cyanobacteria can also be problematic because of the harmful toxins they can produce. Cyanobacterial toxins can be either found inside of the cell, “intracellular,” or outside the cell, “extracellular”. The cyanotoxins can cause adverse impacts including death not only to aquatic organisms that come in direct contact with them but also to livestock, domestic animals, waterfowl, and in some cases to humans (WHO 1999). These toxins belong to a very diverse group of chemical substances that can be classified according to their biological effects on the systems and organs that they affect most strongly including: hepatotoxins, neurotoxins, cytotoxins, irritants and gastrointestinal toxins, and other cyanotoxins whose toxicological or ecotoxicological profile is still only partially known (Funari and Testai 2008).

Additionally, eutrophication can lead to an increase in organic carbon (both particulate and dissolved) in the waterbody. This additional carbon can result in the increased formation of trihalomethanes, a dangerous by-product arising from disinfection of the waterbody with a halogen-containing compound. The chlorine or bromine in the disinfectant reacts with organic compounds in the water to form these tri-substituted methanes that are considered environmental pollutants and are often carcinogenic to humans. Currently, the EPA has set the limit of this total trihalomethanes at 80 ppb in treated water (USEPA 1998).

Common water treatment methods include filtration, softening, reverse osmosis, chlorination, and distillation (Parsons and Jefferson 2006). Other, more specialized treatments include using powdered activated carbon (PAC) to absorb pollutants, and ozonation. Using PAC treatment requires activated carbon to be added to an aerobic or anaerobic treatment system where it serves as the “buffer” against toxics in the wastewater. To treat geosmin and MIB with this method typically requires large doses of activated carbon, and thus can be impractical, particularly for large facilities. Ozonation uses radical formation to eliminate organic particles via coagulation or chemical oxidation. However, like PAC treatment, this method is expensive especially for large treatment facilities; it also can add harmful bi-products to the water. Ozonation has stronger germicidal properties than chlorination however because it rapidly reacts with bacteria, viruses, and protozoans.

The city of Waco, Texas is constructing a water clarification facility as part of the second phase of the Water Quality and Quantity project as a result of a comprehensive five-year study of Lake Waco which identified a significant cyanobacteria population that was dominating the lake as a result of nutrient loading from the watershed. This abundance of blue-green algae further exacerbated the pre-existing taste and odor issues that the reservoir already faced. This new facility, with a construction cost of $40 million dollars, proposes to improve the taste of the finished water by implementing Dissolved Air Flotation (DAF) technology (Waco Water Utilities Services 2009). DAF works by attaching air bubbles to particles suspended in water and floating them to the surface for tank collection. This technology has proven successful in other systems and is particularly effective in waters with a significant amount of lightweight particles (such as algae).

The effectiveness of a specific drinking water treatment for the removal of cyanobacterial toxins depends on the total concentration of the algal toxins, the form of the toxin, and whether it is intracellular or extracellular (Westrick 2008). Intracellular cyanotoxins can be removed by conventional treatment and membrane filtration. Extracellular cyanobacterial toxins are more costly to remove because conventional treatment (e.g., flocculation, coagulation, sedimentation and filtration) is usually not effective and advanced treatment processes must be implemented unless the contaminant is oxidized through disinfection (Westrick 2008).

Due to data limitations, EPA was unable to estimate changes in drinking water treatment costs associated with improvements in taste and odor of source waters resulting from reduced sediment and nutrient discharges from construction sites.

9.2 Avoided Costs of Drinking Water Treatment from Reducing Sediment Discharge from Construction Sites

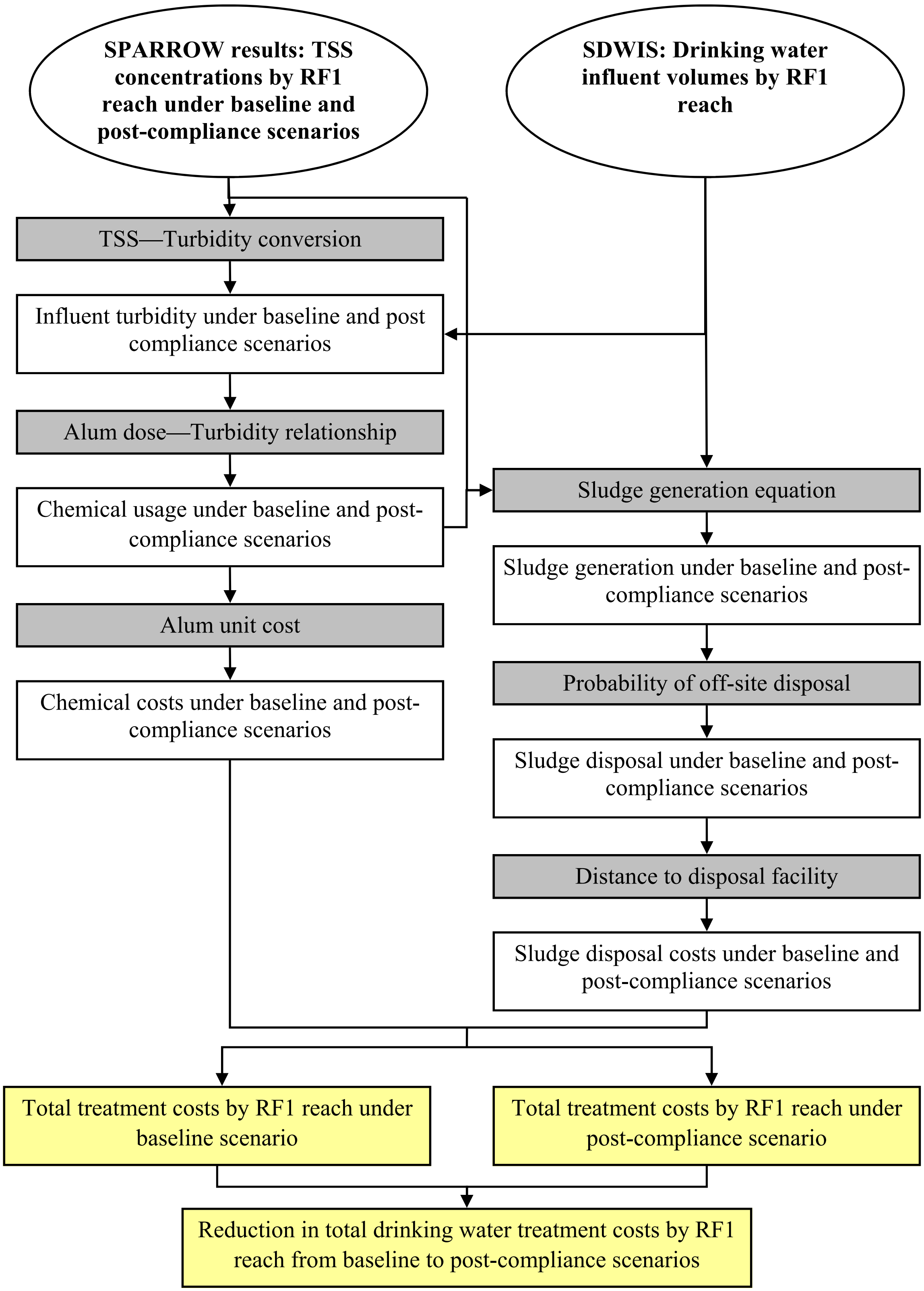
This section describes the estimation of the total expenditures to remove sediments from drinking water and the avoided costs expected with the reduction of sediment discharge anticipated from the regulation. It involves the following steps:

- Identifying RF1 reaches modeled by SPARROW that are sources for drinking water treatment plants
- Determining TSS reductions in these reaches
- Estimating the chemical cost of treating the turbidity caused by TSS in these reaches
- Estimating the cost of disposing of the sludge generated from this turbidity treatment
- Estimating the total costs of drinking water treatment
- Estimating the total avoided costs between the baseline and post-compliance scenarios.

Error! Reference source not found. outlines these steps in more detail. The ovals at the top represent primary data sources. The gray boxes contain calculations or conversions, and the white boxes contain the results of these calculations or conversions that serve as input for the next step in the calculation. The left-hand side of the diagram shows the calculation of the chemical cost of treating turbidity, while the right-hand side shows the calculation of the cost of sludge disposal. Finally, the pale yellow boxes at the bottom of the figure represent the costs under baseline and post-compliance scenarios and the difference between them.

Sediment concentrations and drinking water influent volumes were estimated for each surface drinking water intake in the United States. EPA calculated the costs of treatment chemicals and sludge disposal by surface drinking water intake; the specific steps are described in more detail in the following sections. Costs under the baseline and post-compliance scenarios were compared, and the difference represents the national avoided costs. To address uncertainty in its assumptions, EPA conducted a sensitivity analysis that varies the assumptions described in the following sections, and the results of the following analyses present low, midpoint, and high benefits estimates (see *Section 9.8* for details).

Figure 9-1: Steps in Calculating the Cost of Treating Turbidity in Drinking Water



9.3 Data Sources

Information on drinking water intake points and volumes were provided by EPA's Safe Drinking Water Information System (SDWIS), Federal Version. The federal version of SDWIS stores the information on approximately 160,000 public water systems used to monitor the quality and safety of drinking water (USEPA 2006b). For this analysis, EPA was only interested in surface water intakes, as these are the only type that would be affected by discharge from construction sites. Location information and average daily flow were estimated from the latitude-longitude and population data provided by SDWIS, respectively.

EPA obtained data on the chemicals used to treat turbidity based on data from the American Society of Civil Engineers (ASCE) and the American Water Works Association (AWWA). These data also contain a formula for the calculation of sludge disposal costs (ASCE/AWWA 2005).

Estimations of coagulant dosing for various turbidity levels were taken from research reports by Cornell University's AguaClara Project. This project focuses on improving drinking water quality through engineering research and offers calculations and other engineering information for both researchers and professionals in the field (Menéndez et al. 2007).

The costs of alum and alternative coagulant inputs were obtained by EPA for the Long Term 2 Enhanced Surface Water Treatment Rule. The *Technologies & Costs Document* for this rule included research on the expenses of public water systems and potential changes resulting from the rule (USEPA 2005c). For this purpose, many details of the costs of drinking water treatment were obtained. This document reports cost data in 1998 dollars, so its data are be adjusted to 2008 dollars.

The equation used to estimate the volume of sludge generated from turbidity and TSS treatment comes from the EPA document *Identification and Characterization of Typical Drinking Water Treatment Residuals* (USEPA 2007a), which examines the byproducts of drinking water purification from different methods of treatment.

Calculations of the percentage of facilities that dispose of any of their residual sludge off-site and of the percentage of total residual sludge generated that is disposed of off-site are obtained from the Drinking Water Treatment Plant Master Survey Response Database (henceforth the Drinking Water Survey Database) prepared by EPA for the Drinking Water Effluent Limitation Guidelines rule (USEPA 2007b).

9.4 Modeling Sediment Concentrations and Reductions

EPA modeled sediment concentrations in RF1 stream reaches under both the current and final scenarios using the SPARROW model. A detailed description of this model can be found in *Chapter 6*.

Drinking water treatment facilities located on waterbodies with typically higher levels of TSS and turbidity may have systems in place that reduce total suspended solids and turbidity levels in influent water before it is chemically treated. EPA assumed, as per Ngo and Nichols (1998), that facilities with baseline TSS concentrations exceeding 1,000 mg/L would employ a process such as plain sedimentation, during which influent water is allowed to settle for extended periods of days or weeks. The effectiveness of this process depends on the size of the sedimentation basin and the amount of time the water is left to settle. Studies by Yao (1973) and DeZuane (1997) suggest that the effectiveness of plain sedimentation ranges from 30 and 90 percent with a midpoint estimate of 60 percent. Thus, EPA varied the effectiveness of plain sedimentation basins between 90, 60, and 30 percent TSS removal for the low, mid, and high benefit estimates, respectively.[12]

This reduction is applied to influent water TSS concentrations at facilities that are estimated to have influent water with TSS concentrations exceeding 1,000 mg/L in the baseline. EPA deems this approach reasonable, as it brings treatment costs within the range of values published in existing literature presented in *Section 9.1*.

9.5 Identifying Reaches with Drinking Water Intakes and Their Intake Volumes

To estimate the avoided costs of drinking water treatment resulting from the regulation, EPA used the drinking water intake database compiled by USGS. The database provides information on public water systems, including intakes, treatment plants, sampling sites, distribution systems, wells, and consecutive connections from EPA's SDWIS database. Each drinking water intake is matched to the RF1 reach and has the corresponding "USGS River Reach Number." The database, however, does not provide information on the intake volume. For this analysis, EPA approximated each intake volume based on the population served by each drinking water system and the number of intakes corresponding to that system. The population served by a particular drinking water system was divided by the number of intakes in that system to obtain the average population served per intake. From these data the daily flow of each intake was estimated using the following equation determined by EPA for the Revised Arsenic Rule (USEPA 2000a):

$$F = 0.067 * P^{1.062} \qquad \text{(Eq. 9-1)}$$

Where:

F = average daily flow in thousands of gallons per day

P = population served

Table 9-2 summarizes public surface drinking water intakes by EPA region and the total estimated daily flow for all intakes in that region. Overall, public water systems in the United States withdraw nearly 18 billion gallons of water per day from surface water sources, an average of 2.8 million gallons per day per intake.

[12] The more effective the plain sedimentation process, the lower the treatment costs to the affected facilities, and consequently, the lower their potential savings from reduced influent turbidity.

Table 9-2: Public Water Intakes and Estimated Average Daily Flow by EPA Region

EPA Region	Number of Public Water Intakes	Total Estimated Daily Flow (million gallons/day)
1	639	1,105
2	478	2,286
3	881	2,301
4	868	2,913
5	525	2,373
6	821	2,309
7	353	715
8	549	701
9	748	2,543
10	487	560
Total	**6,349**	**17,809**

Source: SDWIS, Federal Version (USEPA 2006c).

9.6 Estimating the Cost of Chemical Turbidity Treatment

Reducing sediment concentrations in drinking water influent is expected to reduce the turbidity level of the water, requiring smaller doses of chemical coagulants to treat. EPA estimated the amount of coagulants needed to treat a given level of turbidity according to the following steps:

- Converting sediment concentrations (given in mg/L) into nephelometric turbidity units (NTU) using an equation determined by ASCE/AWWA (2005):

$$T = \frac{TSS}{b} \qquad \text{(Eq. 9-2)}$$

Where:

T = turbidity (in NTU)

TSS = TSS (in mg/L)

b = 0.8, 1.5, and 2.2 for the low , midpoint and high estimates,(as the value of *b* decreases, a given level of TSS generates more turbidity, leading to higher treatment costs).

- Determining the doses of alum (the primary coagulant used to treat turbidity) required to treat this turbidity level using relationships determined by researchers of the AguaClara Project (Menéndez et al. 2007).
- Multiplying this coagulant dose (given per volume of water treated) by the total volume of a drinking water intake to obtain the total coagulant usage for water from that intake.
- Multiplying this amount of coagulant by the unit cost of the alum.

For this analysis, EPA assumed that alum is the coagulant used to treat turbidity, since ferrous sulfate is likely to be used only on water with low turbidity (high doses of iron may have negative effects on water quality). No cost information was available for ferrous sulfate, but ferric chloride was quoted at $497 per ton (2008$); additionally, dosing information for iron compounds could not be determined. For the Long Term 2 Enhanced Surface Water Treatment Rule, EPA determined that alum costs between $286 and $373 per ton (2008$) (USEPA 2005c). EPA's assumption that all turbidity will be treated with alum may understate treatment costs and savings. EPA estimated avoided costs using the cost of both dry stock and liquid stock alum to present a range of savings estimates in the sensitivity analysis. The lower cost of liquid alum was used in the low estimate, but it should be noted that the sludge generation equation in *Section 9.7.1,* calculates sludge volume based on dry alum as a coagulant. Therefore, the cost of dry alum is used for the midpoint and high estimates. The approximate dose of alum used to treat water of a given turbidity can be estimated using the following equation:

$$Al = 33\log(T) - 28 \qquad \text{(Eq. 9-3)}$$

Where:

Al = alum dose in milligrams per liter

T = influent water turbidity in NTU (after any reduction from plain sedimentation).

After calculating the dose of alum necessary to treat the turbidity in the influent water at a given intake, this dose was multiplied by the total daily influent volume of the intake (converted from gallons to liters using a conversion factor of 3.8 liters to one gallon) to obtain the total amount of alum needed to treat turbidity at the intake per day. This total daily amount of alum was multiplied by 365 to obtain an annual estimate of the amount of alum needed to treat turbidity from the intake. EPA then used a conversion factor of $907\text{x}10^6$ milligrams per ton to estimate the amount of alum required for turbidity treatment in tons per year. Finally, the estimated annual amount of alum was multiplied by the unit cost of alum to arrive at a final annual cost of alum for each drinking water intake. The following equation presents this calculation of the annual cost of alum.

$$TC_{Al} = Q * \left(\frac{Al * 3.8}{907 * 10^6} \right) * 365 * C_{Al} \qquad \text{(Eq. 9-4)}$$

Where:

TC_{Al} = total annual alum cost in 2008$

Q = daily plant flow in gallons

Al = alum dose in milligrams per liter

3.8 = liters in one gallon

$907*10^6$ = milligrams in one ton

365 = days in one year

C_{Al} = cost of one ton of alum in 2008$

The estimated annual cost of turbidity treatment was calculated under the baseline and post-compliance scenarios for each surface drinking water intake in the SDWIS database where TSS concentration information was available, a total of 6,132 intakes.

9.7 Estimating the Cost of Sludge Disposal

The alum added to the water to treat turbidity binds to the particles of sediment, making heavier lumps of coagulated sediment that settle out of the water column. This residual material, known as sludge, is proportional to the amount of sediment in the water and the amount of coagulant added. Drinking water treatment facilities must regularly dispose of this sludge, either by discharging it to a receiving waterbody or wastewater treatment plant, or by having it hauled away for off-site disposal. The cost of sludge disposal can be significant.

The estimation of the cost of sludge disposal has several steps:

- Estimating the amount of sludge generated
- Calculating the probability that a generated quantity of sludge will be disposed of off-site (as opposed to discharged to a receiving waterbody)
- Estimating the distance to the disposal location
- Calculating the cost of sludge disposal

9.7.1 Estimating the Amount of Sludge Generated

A reduction in the turbidity of the water will result in a reduction in the amount of sludge generated from the chemicals added to treat turbidity, and thus a reduction in the cost of sludge disposal. Sludge quantities are a function of the amount of chemical coagulants added to treat the water and the TSS levels of the water. EPA (USEPA 2007c) derived the following relationship between sludge generation and chemical coagulants:

$$S = 8.34 * Q * [2.0CaCH + 2.6MgCH + CaNCH + 1.6MgNCH + CO_2 + 0.44Al + 2.9Fe + SS + A] \quad \text{(Eq. 9-5)}$$

Where:

S = sludge production (lb/day)

Q = plant flow (mgd)

$CaCH$ = calcium hardness removed as $CaCO_3$ (mg/L)

$MgCH$ = magnesium hardness removed as $CaCO_3$ (mg/L)

$CaNCH$ = noncarbonated calcium hardness removal as $CaCO_3$ (mg/L)

$MgNCH$ = noncarbonated magnesium hardness removed as $CaCO_2$ (mg/L)

CO_2 = carbon dioxide removed by lime addition, as $CaCO_3$

Al = dry alum dose (mg/L) (as 17.1 percent Al_2O_3)

Fe = iron dose as Fe (mg/L)

SS = raw water suspended solids (mg/L)

A = other additives (mg/L)

EPA used this equation in conjunction with the estimated sediment concentrations in the source water, the quantity of water taken up by the drinking water intake, and the alum doses required to treat turbidity to estimate the amount of sludge generated from turbidity treatment.

EPA recognizes that the estimated volume of sludge takes into account the residuals generated from alum treatment of TSS, and thus does not represent the actual volumes of sludge generated from the entire water treatment process. Leaving out the other chemical additives does not affect the difference in sludge generation from turbidity treatment under the baseline and post-compliance scenarios, as they would be held constant under both scenarios.

9.7.2 Calculating the Probability That the Sludge Will Require Off-Site Disposal

Public water systems may use different sludge disposal methods. Some facilities dispose of their sludge off-site; some discharge to water bodies under an NPDES permit or discharge to a wastewater treatment plant. Facilities may also discharge a certain amount of their residual sludge and dispose of the remainder off-site. Facilities discharging sludge into receiving waters or to a wastewater treatment plant do not incur sludge disposal costs. To calculate the amount of sludge disposed of off-site and resulting in disposal costs to a drinking water treatment facility, EPA calculated the probability of off-site disposal.

The Agency used facility data from the drinking water survey database to calculate the number of facilities reporting any off-site disposal and the percent of sludge being disposed of off-site at these facilities. The survey found 207 out of 511 facilities (41 percent) reporting some off-site disposal. At these facilities, on average 85 percent of all residual solid sludge was disposed of off-site. To determine the likelihood that a generated quantity of sludge will result in off-site disposal costs for a facility, EPA multiplied the probability that the sludge was generated at a facility performing any off-site disposal (41 percent) by the probability that a unit of sludge from this facility will be disposed of off-site (85 percent). The estimated probability *(Pr)* that a given volume of sludge generated will be disposed of off-site, thus incurring sludge disposal costs, is 35 percent.

9.7.3 Estimating the Distance to the Disposal Location

The cost of sludge disposal is also a function of the distance to the disposal location. A sample of 10 facilities and their disposal locations was taken randomly from the Drinking Water Survey Database. Because location information was available for both the facilities and their disposal locations, EPA calculated the driving distance between the facility and its discharge locations for each of the 10 facilities. If a facility had more than one listed discharge location, an average distance was used. Based on this sample of 10 facilities, the distance to a disposal location can vary between 1 and 200 miles, and the estimated average distance *(Z)* to disposal locations is 21.6 miles.

9.7.4 Calculating the Cost of Sludge Disposal

EPA estimated the annual cost of sludge disposal as a function of sludge volumes and distance to the disposal facility using the following equation from EPA (USEPA 1996a):

$$Y = \left[54.12\left(S * 365 * .0005 * \Pr\right)\right] + \left[(3.83 + 0.25Z)\left(S * 365 * .0005 * \Pr\right)\right] \qquad \text{(Eq. 9-6)}$$

Where:

Y	=	annual cost of sludge disposal ($/year)
54.12	=	disposal cost per ton of sludge ($)
S	=	total sludge production from treating turbidity in drinking water (lbs/day)
365	=	days per year
0.0005	=	tons per lb
Pr	=	the estimated probability of off-site sludge disposal (35 percent)
3.83	=	fixed transportation cost per pound of sludge ($)
0.25	=	variable transportation cost per mile per pound of sludge ($)
Z	=	average transportation distance of 21.6 miles.

The cost of sludge disposal was estimated under both the baseline and post-compliance scenarios for all surface drinking water intakes in SDWIS for which TSS concentrations were available.

9.8 Sensitivity Analysis

To account for uncertainty in drinking water treatment costs EPA varied several of the parameters of this analysis, including the effectiveness of pre-sedimentation, the amount of turbidity generated by a unit of TSS, and the cost of alum, to provide a range of benefits estimates. The values of these inputs in each estimate range are as follows:

Low Estimate:

- TSS levels exceeding 1,000 mg/L in the baseline are reduced by 90 percent for all options
- TSS = 2.2 * Turbidity
- C_{Al} = $250

Midpoint Estimate:

- TSS levels exceeding 1,000 mg/L in the baseline are reduced by 60 percent for all options
- TSS = 1.5 * Turbidity
- C_{Al} = $327

High Estimate:

- TSS levels at exceeding 1,000 mg/L in the baseline are reduced by 30 percent for all options
- TSS = 0.8 * Turbidity
- C_{Al} = $327

9.9 Estimating the Total Costs of Drinking Water Treatment Under Different Policy Scenarios

EPA estimated the total national costs of treating drinking water for turbidity by summing the estimated annual costs of chemical coagulants added to reduce turbidity and the estimated annual cost of disposing of the sludge resulting from this chemical treatment. *Table 9-3* presents annual estimated total drinking water treatment costs for the baseline scenario, including low, midpoint, and high estimates for avoided costs. Estimated drinking water treatment costs for TSS and turbidity for facilities included in EPA's analysis range from $343 to $651 million nationwide under the baseline scenario, with a midpoint value of $521 million.

Table 9-3: Drinking Water Turbidity Treatment Costs Under the Baseline Scenario

EPA Region	Average Treated Turbidity (NTU)			Treatment Cost (millions of 2008$)		
	Low	Mid	High	Low	Mid	High
1	26.7	40.1	76.9	$6.6	$10.5	$14.9
2	50.3	84.4	177.9	$21.3	$33.9	$44.7
3	71.7	105.6	198.9	$45.5	$61.6	$72.8
4	110.8	163.3	307.7	$60.0	$82.5	$98.2
5	136.8	263.8	613.3	$52.6	$83.6	$105.8
6	151.5	367.2	960.3	$60.3	$90.1	$110.8
7	166.0	422.9	1,129.5	$18.4	$33.9	$44.5
8	121.8	254.5	619.4	$18.3	$25.5	$30.3
9	123.8	256.3	620.5	$54.4	$91.5	$119.1
10	43.9	64.4	120.8	$5.1	$7.8	$10.2
Total	**104.0**	**205.9**	**486.0**	**$342.7**	**$520.9**	**$651.3**
Cost per million gallons[1]				**$58.4**	**$88.7**	**$110.9**

[1] Calculated for the estimated 17,373 million gallons per day of intake for public water systems in SDWIS.

It is important to note that these avoided cost estimates do not take into account any treatment or disposal avoided costs to public wastewater treatment facilities to which drinking water treatment facilities may route their sludge, nor any environmental benefits from reduced volumes of sludge discharged directly into surface waters by direct drinking water treatment facilities. Therefore, the total benefits of reducing turbidity in drinking water treatment are understated.

9.10 Estimating Avoided costs from Lower Sediment and Turbidity in Drinking Water Influent

The total avoided costs from lowered turbidity resulting from lower TSS concentrations in drinking water influent were estimated as the difference between drinking water turbidity treatment under the baseline and post-compliance scenarios. *Table 9-4*, *Table 9-5*, *Table 9-6*, and *Table 9-7* present estimated savings in drinking water treatment costs from lowered turbidity for the three policy options EPA considered, as well as EPA's final policy option. The tables include low, midpoint, and high estimates for savings under each of these options.

The estimated savings from reduced TSS and turbidity treatment at drinking water facilities are between $978,400 and $2.1 million, varying between the least and most stringent policy options and between the low and high estimates. Region 5 benefits the most from the TSS reductions expected from this regulatory action. The avoided costs in Region 5 under Option 1 account for more than half of the total national savings. Other regions receive a larger portion of benefits under Options 2 through 4, though Region 5 still accounts for the greatest proportion of the total cost savings, though Regions 4 and 6 also show significant savings under Options 2 through 4.

Option 1, which requires non-numeric effluent limitations for all sites, is EPA’s least stringent policy option. Average turbidity reductions are less than 1 NTU for this policy option, and the estimated savings drinking water treatment costs range between $978,400 and $1.3 million, with a midpoint estimate of $1.2 million.

Option 2 requires ATS on sites with more than 30 acres disturbed at one time and imposes a 13 NTU turbidity standard while requiring non-numeric effluent limitations on all sites and is similar to the option EPA proposed previously. This option reduces turbidity between 0.4 and 1.3 NTU, translating to $1.4 million to $1.9 million in avoided costs, with a midpoint estimate of $1.8 million.

Option 3, EPA’s most stringent policy option, requires ATS on sites with more than 10 acres disturbed at one time, imposes a 13 NTU turbidity standard on these sites, and requires non-numeric effluent limitations on all sites. This option reduces treated turbidity by an average of 0.7 NTU in the midpoint, ranging from 0.4 to 1.4 between the low and high estimates. Total estimated avoided costs for this option are between $1.7 and $2.1 million, with a midpoint estimate just below $2.1 million.

Option 4, requires passive treatment systems on all sites with 10 or more acres disturbed at one time, and establishes a numeric turbidity standard of 250 NTU (expressed as a daily maximum value) for sites required to implement passive treatment. In addition, all sites will be required to meet non-numeric effluent limitations. National average turbidity reductions from Option 4 range from 0.4 NTU to 1.3 NTU, with a midpoint estimate of 0.6 NTU. Total avoided costs for Option 4 range between $1.5 and $1.9 million, with a midpoint estimate of $1.8 million. While Option 4 is less stringent than Option 3, it is estimated to reduce turbidity in drinking water sources by nearly as much and to produce similar monetized benefits.

As construction site discharge is more likely to contain smaller particles that contribute less to TSS and more to turbidity, the high estimates for Options 2, 3, and 4 may be more relevant because EPA uses a conversion factor between TSS and turbidity that takes this into account. The high estimate for turbidity reductions under Option 2 is 1.3 NTU nationwide. Under Option 3, the high estimate is on average around 1.4 NTU nationwide. For Option 4, the selected option, nationwide reductions are 1.4 NTU. In Region 5, high estimates of reductions under all options are 4.1 NTU.

Table 9-4: Reduction in Drinking Water Treatment Costs Under Option 1

EPA Region	Average Reduction in Treated Turbidity (NTU) [1]			Avoided costs (thousands of 2008$)		
	Low	Mid	High	Low	Mid	High
1	0.0	0.0	0.0	$4.0	$5.0	$5.8
2	0.1	0.5	1.8	$44.1	$91.2	$129.8
3	0.0	0.0	0.1	$8.0	$9.2	$9.3
4	0.4	0.5	1.0	$136.2	$161.1	$164.5
5	1.2	1.9	3.9	$515.0	$603.4	$616.6
6	0.3	0.5	1.1	$162.4	$194.1	$201.0
7	0.1	0.2	0.4	$11.3	$14.1	$15.8
8	0.0	0.0	0.0	$1.1	$1.3	$1.4
9	0.1	0.1	0.2	$7.6	$9.9	$11.3
10	0.1	0.2	0.3	$88.7	$110.4	$110.4
Total	**0.2**	**0.4**	**0.9**	**$978.4**	**$1,199.8**	**$1,265.9**

[1] Average turbidity reductions shown as 0.0 are not actually zero, but not sufficiently large to show at this level of significant digits.

Table 9-5: Reduction in Drinking Water Treatment Costs Under Option 2

EPA Region	Average Reduction in Treated Turbidity (NTU) [1]			Avoided costs (thousands of 2008$)		
	Low	Mid	High	Low	Mid	High
1	0.0	0.0	0.1	$8.2	$10.2	$11.7
2	0.1	0.6	1.8	$50.8	$99.4	$138.4
3	0.1	0.1	0.2	$24.0	$27.7	$27.8
4	0.7	1.0	1.9	$258.9	$307.6	$318.3
5	1.2	2.0	4.1	$526.6	$618.4	$633.7
6	0.7	1.1	2.3	$341.5	$408.3	$422.5
7	0.2	0.4	0.8	$23.5	$29.3	$33.0
8	0.0	0.0	0.1	$2.2	$2.7	$2.9
9	0.1	0.2	0.4	$15.8	$20.5	$23.3
10	0.2	0.3	0.6	$195.9	$244.9	$244.9
Total	**0.4**	**0.6**	**1.3**	**$1,447.4**	**$1,769.1**	**$1,856.6**

[1] Average turbidity reductions shown as 0.0 are not actually zero, but not sufficiently large to show at this level of significant digits.

Table 9-6: Reduction in Drinking Water Treatment Costs Under Option 3

EPA Region	Average Reduction in Treated Turbidity (NTU)			Avoided costs (thousands of 2008$)		
	Low	Mid	High	Low	Mid	High
1	0.0	0.1	0.1	$10.2	$12.7	$14.6
2	0.1	0.6	1.8	$54.1	$103.7	$143.2
3	0.1	0.1	0.2	$31.8	$36.6	$36.7
4	0.8	1.2	2.2	$312.7	$370.7	$383.2
5	1.2	2.0	4.1	$532.4	$626.4	$643.0
6	0.9	1.4	2.8	$428.6	$513.1	$531.2
7	0.3	0.5	1.0	$29.3	$36.6	$41.2
8	0.0	0.0	0.1	$2.7	$3.4	$3.7
9	0.1	0.2	0.5	$20.1	$26.9	$31.4
10	0.3	0.4	0.7	$237.0	$320.3	$320.3
Total	**0.4**	**0.7**	**1.4**	**$1,658.9**	**$2,050.5**	**$2,148.5**

Table 9-7: Reduction in Drinking Water Treatment Costs Under Option 4

EPA Region	Average Reduction in Treated Turbidity (NTU)			Avoided costs (thousands of 2008$)		
	Low	Mid	High	Low	Mid	High
1	0.0	0.0	0.1	$6.8	$8.5	$9.7
2	0.1	0.6	1.8	$50.3	$98.8	$137.8
3	0.1	0.1	0.2	$25.6	$29.6	$29.6
4	0.7	1.1	2.0	$281.1	$333.5	$344.8
5	1.2	2.0	4.1	$526.8	$618.8	$634.2
6	0.8	1.3	2.6	$396.9	$474.5	$490.6
7	0.2	0.4	0.9	$27.0	$33.7	$37.8
8	0.0	0.0	0.1	$2.2	$2.8	$3.1
9	0.1	0.2	0.4	$16.4	$21.5	$24.7
10	0.2	0.2	0.4	$145.5	$181.9	$181.9
Total	**0.4**	**0.6**	**1.3**	**$1,478.7**	**$1,803.5**	**$1,894.3**

9.11 Sources of Uncertainty/Limitations

Key limitations of EPA's analysis of avoided costs to drinking water treatment facilities are outlined below. The SPARROW model for suspended sediments that was used for estimating TSS concentrations in the source waters also has a number of limitations. These limitations are discussed in detail in *Chapter 6*.

- EPA's analysis of reductions in TSS and turbidity in surface waters is limited to the RF1 reaches modeled by SPARROW. Therefore, reductions at any facilities that are not located on RF1 reaches would not be included in this assessment of benefits to drinking water treatment, leading to an underestimation of benefits.
- Sediment filtration systems and pre-sedimentation (allowing water to sit and sediment to filter out before treatment) at drinking water treatment facilities reduce the sediment concentration of the water before it enters chemical treatment, so that the turbidity level of

the water entering the facility is not the turbidity level that is eventually treated with coagulants. Assuming that the differential between pre- and post- compliance sediment concentration is proportional to the differential between pre- and post-compliance turbidity treatment introduces uncertainty, as the lower sediment levels may be more or less affected by the pre-sedimentation and filtration processes. EPA's analysis attempts to account for this uncertainty by varying the maximum amount of TSS and turbidity treated by a given drinking water treatment facility.

- If a drinking water treatment facility produces sludge that is toxic (due to other pollutants in the water besides sediment), its disposal costs may be significantly higher because toxic sludge disposal is more restricted and costly. If the facility cannot separate the sludge generated by sediment treatment from the sludge generated by treatment of toxics (which is likely the case), then all of its sludge will be characterized as toxic. This analysis may understate the cost of disposal (and thus the avoided costs of smaller quantities of sludge to be disposed of) for facilities that generate toxic sludge.
- Reducing nutrient loadings to surface waters is expected to reduce eutrophication which is one of the main causes of taste and odor impairment in drinking water. Taste and odor in drinking water has a major negative impact on the public perception of drinking water safety and the drinking water industry due to a significant increase in drinking water treatment costs from foul taste and odor in the source waters. Although the final regulation is expected to reduce the cost of drinking water treatment to improve taste and odor, the Agency was unable to monetize this benefit category due to data limitations.
- The analysis presumes that reduced costs of drinking water treatment will not result in appreciable changes in prices (costs) of drinking water paid by households. Given the marginal costs of reduced turbidity treatment relative to other costs of drinking water production, this assumption seems valid. However, if substantial price effects do occur, additional benefits will accrue at the consumer level.

10 Nonmarket Benefits from Water Quality Improvements

As discussed in the preceding chapters of this document, sediments, nutrients, and other pollutants from construction sites may have a wide range of effects on water resources located in the vicinity of the construction sites. These environmental changes affect environmental services valued by humans, including recreation; commercial fishing; public and private property ownership; navigation; water supply and use; and existence services such as aquatic life, wildlife, and habitat designated uses. These nonmarket benefits (Freeman 2003) are in addition to market benefits (e.g., avoided costs of producing various market goods and services) estimated in prior chapters.

This chapter describes the use of meta-analysis of surface water valuation studies for estimating benefits of water quality improvements resulting from the regulation. The technical details involved in the estimation of meta-analysis are presented in *Appendix G* as well as in sources such as Bateman and Jones (2003), Johnston et al. (2005, 2006), Shrestha et al. (2007), and Rosenberger and Phipps (2007).

10.1 Water Quality Index

To link water quality changes from reduced sediment discharge to effects on human uses and support for aquatic and terrestrial species habitat, this analysis utilizes a water quality index (WQI). The index translates water quality measurements, gathered for multiple parameters that are indicative of various aspects of water quality, into a single numerical indicator. The parameters used in formulating the WQI are determined based on waterbody type, scientific understanding of ecosystem response to varying conditions, and available data.

Most importantly for the present analysis, the WQI provides the link between specific pollutant levels, as reflected in individual index parameters (e.g., dissolved oxygen concentrations), and the presence of aquatic species and suitability for particular recreational uses. The WQI value, which is measured on a scale from 0 to 100, reflects varying water quality, with 0 for poor quality and 100 for excellent.

Numerous water quality indices have been developed and documented in the literature since the 1960s (Horton 1965); however, no standardized approach has emerged. EPA Region 10 was one of the first designers of the WQI framework in use today (Brown et al. 1970). The National Sanitation Foundation WQI (McClelland 1974) and Oregon WQI (Dunnette 1979) proposed subsequent iterations that built on this original framework. McClelland (1974) relied on a survey of water quality experts to identify water quality parameters to be included in the index, the structure and the relationship between the parameters, and the weights given to the parameters in the WQI equation. This is a procedure known as the Delphi Method (Dalkey 1968).

The WQI used in this analysis builds on McClelland's work, and the methodology developed by Dunnette (1979), which was subsequently updated by Cude (2001) to better account for spatial and morphologic variability in the natural characteristics of streams. To support the analysis of changes in sediment loadings from construction sites relative to baseline conditions, EPA adapted Cude's methodology to the national scale by developing subindex curves for TSS at the level of ecoregions. EPA also further modified the WQI methodology to apply to estuarine reaches, building on prior applications of WQI concepts to saltwater environments (Harrison et al. 2000; Canadian Council of Ministers of the Environment 2001; Carruthers and Wazniak 2003; Gupta et al. 2003; and USEPA 2007a).

Implementing the WQI methodology involves three key steps: (1) obtaining water quality measurements for each parameter included in the WQI; (2) transforming measurements to subindex values expressed on a common scale; and (3) weighting and aggregating the individual parameter subindices to obtain an overall WQI value that reflects waterbody conditions across the parameters.

In compiling water quality data for each parameter, EPA used a combination of average water quality field measurements and SPARROW modeling results as input values for each of the parameters included in the WQI. Data sources for measurements are discussed in *Section 10.1.4*, and SPARROW modeling results were discussed in *Chapter 6*.

The second step in the implementation of the WQI involves the transformation of parameter measurements into subindex values that express water quality conditions on a common scale of 0 to 100. With the exception of the TSS parameter, EPA used the subindex transformation curves developed by Dunnette (1979) and Cude (2001) for the Oregon Water Quality Index. In the case of TSS concentrations, EPA adapted the approach developed by Cude (2001) to account for the wide range of natural or background sediment concentrations that result from varying geologic and other region-specific conditions, and to reflect the national context of the analysis. TSS subindex curves were developed for each Level III ecoregion (USEPA 2009d) using baseline TSS concentrations calculated in SPARROW at the RF1 reach level.[13] For each of the 85 Level III ecoregions intersected by the RF1 reach network, EPA derived the TSS transformation curve by assigning a score of 100 to the 25th percentile of the reach-level TSS concentrations in the ecoregion (i.e., using the 25th percentile as a proxy for "reference" concentrations), and a score of 70 to the median concentration. An exponential equation was then fitted to the two concentration points.

The final step in implementing the WQI involves combining the individual parameter subindices into a single WQI value that reflects the overall water quality across the parameters. Following McClelland's approach, EPA calculated the overall WQI for a given reach using a weighted geometric mean function as follows:

$$WQI_r = \prod_{i=1}^{n} Q_i^{Wi} \qquad \text{(Eq. 10-1)}$$

Where:

WQI_r	=	the multiplicative water quality index (from 0 to 100) for reach *r*
Q_i	=	the water quality subindex measure for parameter *i*
W_i	=	the weight of the *i-th* parameter
n	=	the number of parameters

The WQI methodology used for rivers and streams and for estuaries are described in *Section 10.1.1* and *Section 10.1.2*, respectively.

13 The selected data exclude outlier TSS concentration values, defined as values that exceed the 95th percentile based on the national population of all RF1 reaches.

10.1.1 Freshwater WQI for Rivers and Streams

The freshwater WQI used by EPA to evaluate water quality in rivers and streams includes six parameters selected to represent major stream impairment categories: dissolved oxygen (DO), biochemical oxygen demand (BOD), fecal coliform (FC), total nitrogen (TN), total phosphorus (TP), and total suspended solids (TSS). These freshwater WQI parameters are based on the set of parameters used in the WQI developed by Cude (2001) but exclude temperature and pH. Temperature and pH are not listed as common causes of impairment nationally or with respect to the uses assessed in this economic valuation.[14] Additionally, temperature may be considered to be indirectly reflected in the DO parameter because DO is highly temperature dependent. In addition, the WQI developed by Cude (2001) does not explicitly account for turbidity associated with water quality impacts from TSS and eutrophication.

Table 10-1 presents parameter-specific functions used for transforming water quality data into water quality subindices for freshwater waterbodies. The equation parameters for each of the 85 ecoregion-specific TSS subindex curves are provided in *Appendix F*.

[14] EPA (2009a) reports 38,228 miles and 26,569 miles of rivers or streams impaired or threatened due to temperature and pH, respectively, as compared to much higher incidence of impairment due to pathogens (112,084 miles), sediment (85,248 miles), and nutrients (81,279 miles). Temperature and pH are similarly responsible for a relatively small share of impairments noted in estuaries and bays.

Table 10-1: Freshwater Water Quality Subindices			
Parameter	**Concentrations**	**Concentration Unit**	**Subindex**
Dissolved Oxygen (DO)			
DO saturation ≤100%			
DO	≤ 3.3	Mg/L	10
DO	3.3 < DO < 10.5	Mg/L	$-80.29+31.88*DO-1.401*DO^2$
DO	10.5 ≤DO	Mg/L	100
100% < DO saturation ≤ 275%			
DO	N/A	Mg/L	100 * exp((DOsat - 100) *-1.197 E-2)
275% < DO saturation			
DO	N/A	Mg/L	10
Fecal Coliform (FC)			
FC	≤ 50	Lbs/100 mL	98
FC	50 < FC ≤ 1,600	Lbs/100 mL	98 * exp(FC - 50) * -9.9178 E-4
FC	> 1,600	Lbs/100 mL	10
Total Nitrogen (TN)			
TN	≤ 3	Mg/L	100 * exp(TN * -0.4605)
TN	> 3	Mg/L	10
Total Phosphorus (TP)			
TP	≤ 0.25	Mg/L	100 - 299.5 * TP - 0.1384 * TP2
TP	> 0.25	Mg/L	10
Total Suspended Solids (TSS)[1]			
TSS	≤ TSS_{10}	Mg/L	10
TSS	TSS_{10} < TSS ≤ TSS_{100}	Mg/L	a * exp(TSS*b); where a and b are ecoregion-specific values
TSS	> TSS_{100}	Mg/L	100
Biochemical Oxygen Demand, 5-day (BOD)			
BOD	≤ 8	Mg/L	100 * exp(BOD * -0.1993)
BOD	> 8	Mg/L	10

[1] TSS_{10} and TSS_{100} are ecoregion-specific TSS concentration values that correspond to subindex scores of 10 and 100, respectively.
Source: Cude (2001) (ecoregion-specific curves were developed for TSS based on Cude's methodology).

Table 10-2 lists the freshwater WQI parameters and the weights used in aggregating the subindex values into the overall WQI for inland rivers and streams. EPA (USEPA 2002) revised the weights originally developed by McClelland (1974) by redistributing the weights to the six parameters retained in the EPA WQI so that the ratio among the parameters is maintained and the weights sum of 1.

Table 10-2: Original and Revised Weights for Freshwater WQI Parameters		
Parameters	**Original Weights (McClelland 1974)**	**Revised Weights (USEPA 2002)**
Dissolved oxygen (% saturation and mg/L)	0.17	0.24
Fecal coliform (colonies/100 mL)	0.16	0.22
Biochemical oxygen demand, 5-day (mg/L)	0.11	0.15
Total nitrogen (mg/L)	0.10	0.14
Total phosphorus (mg/L)	0.10	0.14
Total suspended solids (mg/L)	0.08	0.11
pH (standard)	0.11	--
Temperature (C)	0.10	--
Total Solids (mg/L)	0.07	--
Total	**1.00**	**1.00**

10.1.2 WQI Methodology for Estuaries

The methodology used in calculating a WQI score for estuaries and coastal waterbodies adapts the approach outlined above for rivers and streams to account for the different characteristics and response of saltwater systems. EPA reviewed several applications of WQI concepts to saltwater systems (Harrison et al. 2000; Carruthers and Wazniak 2003; Gupta et al. 2003; and USEPA 2007a) to identify water quality parameters pertinent to estuaries and their relative importance in characterizing overall water quality.

The estuarine WQI includes parameters that are common to both freshwater and saltwater systems—dissolved oxygen, nitrogen, phosphorus, fecal coliform, and suspended solids—and replaces biochemical oxygen demand with chlorophyll-a (ChA). Chlorophyll-a assesses the amount of phytoplankton present in the waterbody. While phytoplankton is a natural component of the waterbody's food web, changes in its abundance (e.g., high ChA), species composition, and productivity are commonly the first biological response to nutrient enrichment (Chesapeake Bay Program 2008). Subsequent decomposition of the phytoplankton may reduce available dissolved oxygen and induce hypoxia and anoxia.

Similarly to the WQI for rivers and streams, the data for each parameter included in the WQI for estuaries are transformed to a standard (0–100) scale using subindex curves. *Table 10-3* presents parameter-specific functions used in transforming water quality data to subindices for estuaries. The curve used for ChA is derived from the curves used for TN, since both parameters indicate trophic status.

Table 10-3: Estuarine Water Quality Subindices

Parameter	Concentrations	Concentration Unit	Subindex
Dissolved Oxygen (DO)			
DO saturation ≤100%			
DO	≤3.3	mg/L	10
DO	3.3<DO<10.5	mg/L	$-80.29+31.88*DO-1.401*DO^2$
DO	10.5≤ DO	mg/L	100
100% < DO saturation ≤ 275%			
DO	N/A	mg/L	$100 * \exp((DO_{sat} - 100) * -1.197 E-2)$
275% < DO saturation			
DO	N/A	mg/L	10
Fecal Coliform (FC)			
FC	≤50	lbs/100 mL	98
FC	50 < FC ≤1,600	lbs/100 mL	98 * exp(FC - 50) * -9.9178 E-4
FC	> 1,600	lbs/100 mL	10
Total Nitrogen (TN)			
TN	≤3	mg/L	100 * exp(TN * -0.4605)
TN	> 3	mg/L	10
Total Phosphorus (TP)			
TP	≤ 0.25	mg/L	$100 - 299.5 * TP - 0.1384 * TP^2$
TP	> 0.25	mg/L	10
Total Suspended Solids (TSS)[1]			
TSS	$\leq TSS_{10}$	mg/L	10
TSS	$TSS_{10} < TSS \leq TSS_{100}$	mg/L	a * exp(TSS*b); where a and b are ecoregion-specific values
TSS	$> TSS_{100}$	mg/L	100

Table 10-3: Estuarine Water Quality Subindices

Parameter	Concentrations	Concentration Unit	Subindex
Chlorophyll-a (ChA)			
ChA	≤ 40	µg/L	100 *exp(ChA * -0.05605)
ChA	> 40	µg/L	10

[1] TSS_{10} and TSS_{100} are ecoregion-specific TSS concentration values that correspond to subindex scores of 10 and 100, respectively.
Source: Cude (2001) (ecoregion-specific curves were developed for TSS based on Cude's methodology).

Table 10-4 shows the weighting scheme used to combine the parameters into an overall WQI. These weights are a modification of the ones used for the freshwater WQI.

Table 10-4: Weights for Estuarine WQI Parameters

Parameters	Estuary WQI Weights[1]
Dissolved oxygen (% saturation and mg/L)	0.26
Fecal coliform (colonies/100mL)	0.25
Total nitrogen (mg/L)	0.15
Total phosphorus (mg/L)	0.15
Total suspended solids (mg/L)	0.11
Chlorophyll-a (mg/L)	0.08
Total	**1.00**

[1] Weights used in the freshwater WQI were redistributed among estuarine WQI parameters to account for the removal of BOD and the addition of ChA.

10.1.3 Relation between WQI and Suitability for Human Uses

Once an overall WQI value is calculated, it can be related to suitability for potential uses. Vaughan (1986) developed a water quality ladder (WQL) that can be used to indicate whether water quality is suitable for various human uses (i.e., boating, rough fishing, game fishing, swimming, and drinking without treatment). Vaughan identified "minimally acceptable parameter concentration levels" for each of the five potential uses. Water quality is deemed acceptable for each use if none of the six parameters exceeds the threshold concentration levels. Vaughan used a scale of zero to 10 instead of the WQI scale of zero to 100 to classify water quality based on its suitability for potential uses. *Table 10-5* presents water use classifications and the corresponding WQL and WQI values.

Table 10-5: Water Quality Classifications

Water Quality Classification	WQL Value	WQI Value[1]
… drinking without treatment	9.5	95
… swimming	7.0	70
… game fishing	5.0	50
… rough fishing	4.5	45
… boating	2.5	25

[1] The WQI value corresponding to a given classification of water quality equals the WQL value multiplied by 10.
Source: Vaughan (1986).

10.1.4 Sources of Data on Ambient Water Quality

For river and streams, EPA used the following data sources to obtain ambient concentrations for the six parameters included in the WQI:

- The SPARROW model outputs provided data for baseline and post-compliance concentrations of total nitrogen, total phosphorus, and total suspended solids as discussed in *Chapter 6*.
- The USGS National Water Information System (NWIS) provided concentration data for three parameters: (1) fecal coliform, (2) dissolved oxygen, and (3) biochemical oxygen demand.
- EPA's Storage and Retrieval (STORET) data warehouse provided additional data on fecal coliform counts and biochemical oxygen demand (USEPA 2008e). The NWIS database did not include data for these two parameters for 88.6 percent and 91.3 percent of the 9,444 RF1 reaches, respectively. To address these data gaps, EPA augmented fecal coliform and biochemical oxygen demand data by adding observations from the STORET data warehouse. The addition of these observations increased the fecal coliform and biochemical oxygen demand data, so that 71.9 percent and 33.8 percent, respectively, of the 9,444 reaches were covered.

Baseline freshwater WQI values are based on baseline data for 60,017 reaches in the coterminous United States.[15] Baseline concentrations for all WQI parameters were available for a total of 9,444 reaches. EPA used a successive average approach to address the data gaps in the remaining freshwater reaches. The approach involves assigning the average of ambient concentrations for a WQI parameter within a hydrologic unit to reaches within the same hydrologic unit with missing data and progressively expanding the geographical scope of the hydrologic unit (HUC8, HUC6, HUC4, and HUC2) to fill in all missing data.[16] This approach assumes that waterbody reaches located in the same watershed generally share similar characteristics. Using this estimation approach, EPA was able to compile water quality data for a total of 60,017 freshwater reaches (including 50,573 reaches estimated from HUC-based averages).

The following sources provided data for estuaries and coastal waterbodies:

- The SPARROW model outputs provided data for baseline and post-compliance concentrations of suspended sediment. Some sediment concentrations were calculated from sediment flux using Dissolved Concentration Potentials as discussed in *Chapter 6*.
- EPA's Environmental Monitoring & Assessment Program, National Coastal Assessment Monitoring Data (EMAP-NCA) database provided data for approximately 3,000 sampling stations in the United States, for all pertinent parameters except fecal coliform (USEPA 2008a).

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15 No ambient concentration data were available for Alaska and Hawaii.

16 Hydrologic Unit Codes (HUCs) are cataloguing numbers that uniquely identify hydrologic features such as surface drainage basins. The HUCs consist of 8 to 14 digits, with each set of 2 digits giving more specific information about the hydrologic feature. The first pair of values designate the region (of which there are 21), the next pair the subregion (total of 222), the third pair the basin or cataloguing unit (total of 352), and the fourth pair the subbasin, or accounting unit (total of 2,262) (USGS 2007a). Digits after the first eight offer more detailed information, but are not always available for all waters. In this discussion, a HUC level refers to a set of waters that have that number of HUC digits in common. For example, the HUC6 level includes all reaches for which the first six digits of their HUC are the same.

- The National Estuarine Research Reserve System database (NERRS) provided data for 25 estuarine reserves, each containing multiple water monitoring stations, for all relevant parameters except fecal coliform (NOAA 2008).
- EPA's STORET data warehouse provided additional data on all relevant parameters, including fecal coliform counts, but excluding chlorophyll-a (USEPA 2008e).
- EPA also obtained data from the following state and local agencies through special requests:
  - The Houston Area Research Center (HARC) is a nonprofit organization that supports the water monitoring data collection for the Texas Commission on Environmental Quality. HARC provided data spanning the period of 1999 to 2005 (Gonzalez 2009).
  - The New Hampshire Department of Environmental Quality collects coastal water quality data in support of its Coastal Program. Data include measurements made from 1999 to 2008 (Hunt 2009).
  - The Southern Carolina Estuarine and Coastal Assessment Program is a collaboration between the South Carolina Department of Natural Resources (SCDNR) and the South Carolina Department of Health and Environmental Control (SCDHEC) providing monitoring and periodic reports on the conditions of the state's estuarine habitats. Data are available for the period of 1999 to 2004 (VanDolah 2009).
  - The Connecticut Department of Environmental Protection, on behalf of the collaboration with EPA on the Long Island Sound Study, conducts a Long Island Sound Water Quality Monitoring Program. In total, 28 stations are monitored each year. Data include measurements made from 1999 to 2008 (Lyman 2009).
  - The Washington Department of Ecology conducts marine water quality monitoring at 85 stations in Puget Sound, Grays Harbor, and Willapa Bay. About 40 stations are monitored each year on a monthly basis, with some stations monitored on a rotating schedule. Data cover the period of 1975 to 2008 (Thom 2009).
  - The Oregon Department of Environmental Quality collects coastal water quality data in support of its Water Quality program. Data include measurements made from 1991 to 2008 (Marxer 2009).

In the case of estuaries and coastal reaches, EPA used a similar approach to estimate missing concentration values based on the average concentrations available for reaches within consecutively coarser HUC levels. Thus, for RF1 reaches where no empirical measurement could be obtained from any of the sources listed above, the parameter value was estimated by averaging the corresponding values for adjacent estuarine/coastal reaches within the same HUC8 watershed. Due to the smaller scale of estuarine waterbodies, EPA used the average of concentration values for HUC8, HUC6, and HUC 4 to estimate average parameter concentrations for reaches where data gaps existed and did not estimate values based on HUC2 averages. Using this approach, EPA compiled ambient water quality data for all WQI parameters for 772 reaches out of the total 2,067 estuarine/coastal reaches in the coterminous United States. Due to data limitations 1,295 coastal reaches were excluded from the analysis.

EPA calculated baseline WQI for all reaches based on modeled, measured, or extrapolated parameter values. The WQI was calculated using the methodology outlined above according to the type of reach (either an inland stream or an estuarine reach). SPARROW identifies estuarine and coastal reaches using a specific flag.[17] As done throughout the analysis (see *Chapter 6*), in calculating WQI values, EPA replaced all outlier TSS concentrations equal or above the 95th percentile with the 95th percentile value (6,157.8 mg/L in the baseline). Based on this threshold, a total of 3,106 reaches are considered outliers. *Table 10-6* shows the distribution of reach miles by WQI value and EPA region for the baseline scenario (existing conditions).

Table 10-6: Percentage of Reach Miles in Coterminous 48 States by WQI Classification for EPA Regions: Baseline Scenario

EPA Region	WQI<25	25≤WQI<50	50<WQI<70	70<WQI
1	0.2%	8.3%	82.7%	8.8%
2	0.5%	13.0%	36.4%	50.1%
3	5.0%	43.4%	45.0%	6.6%
4	0.8%	35.8%	42.2%	21.2%
5	6.1%	63.4%	27.1%	3.3%
6	1.0%	64.6%	28.1%	6.3%
7	17.9%	70.9%	10.2%	1.1%
8	5.0%	51.2%	24.8%	18.9%
9	19.0%	73.2%	7.3%	0.6%
10	0.1%	19.2%	35.5%	45.2%
National Average	5.6%	50.3%	29.0%	15.1%

Note: Data include both freshwater and saltwater reaches.

10.1.5 Estimated Changes in Water Quality (ΔWQI) from the Regulation

To estimate benefits of water quality improvements expected from the regulation, EPA calculated the WQI for each policy option. In calculating the post-compliance WQI, the Agency used option-specific TSS, TN, and TP concentrations based on SPARROW output as described in *Chapter 6*. The sediment and nutrient concentration estimates for each policy option reflect the expected reduction in sediment discharge and associated nutrients under the policy options. Although reductions in sediment and nutrient discharges from construction sites may also reduce loadings of other pollutants that are included in the WQI, the other parameters were held constant in this analysis for all policy options.

Each RF1 reach that has an improved WQI value relative to its baseline WQI value contributes to the estimated economic benefits of a policy option. Based on the estimated WQI value under the baseline scenario, EPA categorized each RF1 reach using four WQI ranges (WQI < 25, 25≤WQI<50, 50≤WQI<70, and 70≤WQI). WQI values of less than 25 indicate that water is not suitable for boating (the recreational use with the lowest required WQI), whereas WQI values greater than 70 indicate that waters are swimmable (the recreational use with the highest required WQI). For each WQI category under the baseline scenario and policy options in a given state, EPA estimated the weighted average WQI using reach miles as weights.

The difference in WQI between baseline conditions and a given policy option (hereafter denoted as ΔWQI) is a measure of the change in water quality attributable to the policy option. To monetize benefits of the regulation, EPA used three ranges of water quality improvements $0.01 < \Delta WQI \leq 0.1$, $0.1 < \Delta WQI \leq 0.5$, and $0.5 < \Delta WQI$. Water quality improvements below 0.01 WQI units are not

17 "Terminal" reaches that terminate into the Great Lakes or along the Canadian border were treated as inland rivers and streams.

calculated because they are so small EPA does not expect any benefits from these improvements. For each combination of the baseline water quality category and the improvement range, the Agency estimated average ΔWQI and the corresponding percentage of total reach miles in the state. In this analysis, the numbers of reach miles improved under each option – i.e., miles that experience a change in the WQI value – differ from the numbers presented in *Chapter 6* which instead looked at absolute changes in TSS, TN, or TP concentrations. Additionally, data presented in this chapter cover all reaches within the RF1 network, including those that do not receive construction discharge directly but may nevertheless see improvement due to changes in water quality in upstream reaches. As noted in *Chapter 6*, when considering all RF1 reaches, including those that do not receive construction sediment discharge directly, the analysis shows a total of 431,074 reach miles improving under Options 1, 2, and 4. These miles account for 68.7 percent of the total RF1 reach network. Option 3 has an even broader effect, reducing TSS concentrations in a total of 472,402 reach miles (75.3 percent of the RF1 reach network).

Table 10-7, *Table 10-8*, *Table 10-9*, and *Table 10-10* summarize changes in ambient water quality resulting from Options 1, 2, 3, and 4 respectively. *Appendix H* provides more detail on water quality improvements by the baseline water quality category.

EPA estimated that Option 1 will not result in significant water quality improvements; this option improves water quality in only 73,054 reach miles[18] or 11.6 percent of the 627,679 miles modeled in the analysis. EPA Region 4 shows the most water quality improvements, with 29,911 reach miles (33.1 percent of reach miles modeled in this region) being affected. Region 6 is expected to have the next largest water quality improvement with 23,355 reach miles being affected under the post-compliance scenario.

EPA's Option 2 is estimated to improve ambient water quality in 112,429 reach miles (17.9 percent) that receive construction discharges nationwide. EPA's Regions 1, 4, and 6 are each estimated to experience improved water quality on 20 percent or greater of RF1 reach miles included in the analysis. EPA's water quality analysis predicts that Regions 4 and 6 will experience the most improvements in terms of both reach miles (45,817 and 31,540) and percentage of total regional reach length (50.7 percent and 33.2 percent). EPA's analysis also indicates that Region 6 would benefit the most in terms of the estimated magnitude of water quality improvements with 3.1 percent of reach miles modeled estimated to improve by greater than 0.5 WQI units. Conversely, EPA Region 2 is estimated to improve the least, with 1,532 reach miles benefiting from higher water quality.

Option 3 yields the most significant results overall with 129,747 reach miles expected to improve under the post-compliance scenario. The estimated scale of improvements ranges from 3.2 percent to 55.2 percent of reach miles in EPA's Region 8 and 4, respectively. EPA estimated that Region 4 will experience the largest water quality improvements, with 49,945 reach miles improved (55.2 percent of the reach miles modeled). EPA's analysis also shows that Region 6 will have the most reach miles (3,487) improving by greater than 0.5 WQI units, while Region 2 will have the lowest water quality improvements with 2,147 reach miles showing any improvement under the post-compliance scenario.

Option 4 generates the second most improvements in water quality when compared to the other three regulatory options. EPA calculates that a total of 113,963 reach miles (18.2 percent of reaches in the analysis) will be improved under this option. As with the other three options, Region 4 is expected to have the most improved reaches with 47,130 reach miles (52.1 percent) experiencing some improvements in water quality. EPA Region 2 is expected to improve the least in terms of reach miles, with 1,438 miles

18 For the purpose of the discussion, "reach miles" refers to both freshwater and estuarine RF1 reaches.

(9.5 percent) improving. EPA Region 8 is expected to have the least improvements in terms of percentage of reach miles improved. Only 1.3 percent (1,739 miles) are expected to improve in the region under Option 4. Region 6 has the most miles with water quality improvements greater than 0.5 WQI units, 3,313 miles.

Table 10-7: Estimated Water Quality Improvements Under Option 1[1]

EPA Region	Baseline Scenario: Reach Miles Modeled	Baseline Scenario: Total Reach Miles in RF1 Network	0.01 < ΔWQI < 0.1: Reach Miles	0.01 < ΔWQI < 0.1: % of Reach Miles Modeled	0.01 < ΔWQI < 0.1: % of Total Reach Miles	0.1 < ΔWQI < 0.5: Reach Miles	0.1 < ΔWQI < 0.5: % of Reach Miles Modeled	0.1 < ΔWQI < 0.5: % of Total Reach Miles	0.5 < ΔWQI: Reach Miles	0.5 < ΔWQI: % of Reach Miles Modeled	0.5 < ΔWQI: % of Total Reach Miles	Total Improved Reaches: Reach Miles	Total Improved Reaches: % of Reach Miles Modeled	Total Improved Reaches: % of Total Reach Miles
1	16,182	18,324	1,696	10.5%	9.3%	31	0.2%	0.2%	29	0.2%	0.2%	1,756	10.9%	9.6%
2	15,140	16,110	415	2.7%	2.6%	0	0.0%	0.0%	0	0.0%	0.0%	415	2.7%	2.6%
3	28,904	33,617	1,539	5.3%	4.6%	114	0.4%	0.3%	0	0.0%	0.0%	1,653	5.7%	4.9%
4	90,435	94,525	26,210	29.0%	27.7%	3,172	3.5%	3.4%	529	0.6%	0.6%	29,911	33.1%	31.6%
5	68,285	71,550	2,931	4.3%	4.1%	132	0.2%	0.2%	4	<0.1%	<0.1%	3,067	4.5%	4.3%
6	95,098	98,681	17,902	18.8%	18.1%	4,227	4.4%	4.3%	1,227	1.3%	1.2%	23,355	24.6%	23.7%
7	60,909	60,909	4,196	6.9%	6.9%	562	0.9%	0.9%	168	0.3%	0.3%	4,926	8.1%	8.1%
8	130,311	130,311	495	0.4%	0.4%	64	0.1%	0.1%	362	0.3%	0.3%	921	0.7%	0.7%
9	54,228	56,492	1,360	2.5%	2.4%	134	0.3%	0.2%	151	0.3%	0.3%	1,646	3.0%	2.9%
10	68,189	69,524	3,541	5.2%	5.1%	1,322	1.9%	1.9%	542	0.8%	0.8%	5,404	7.9%	7.8%
Nation	**627,679**	**650,043**	**60,285**	**9.6%**	**9.3%**	**9,757**	**1.6%**	**1.5%**	**3,012**	**0.5%**	**0.5%**	**73,054**	**11.6%**	**11.2%**

[1] Reach miles include both freshwater and estuarine RF1 reaches.

Table 10-8: Estimated Water Quality Improvements Under Option 2[1]

EPA Region	Baseline Scenario		Water Quality Improvements by WQI Change											
			0.01 < ΔWQI < 0.1			0.1 < ΔWQI < 0.5			0.5 < ΔWQI			Total Improved Reaches		
	Reach Miles Modeled	Total Reach Miles in RF1 Network	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles
1	16,182	18,324	3,035	18.8%	16.6%	235	1.5%	1.3%	41	0.3%	0.2%	3,310	20.5%	18.1%
2	15,140	16,110	1,527	10.1%	9.5%	5	<0.1%	<0.1%	0	0.0%	0.0%	1,532	10.1%	9.5%
3	28,904	33,617	4,483	15.5%	13.3%	108	0.4%	0.3%	30	0.1%	0.1%	4,621	16.0%	13.7%
4	90,435	94,525	37,936	42.0%	40.1%	6,594	7.3%	7.0%	1,288	1.4%	1.4%	45,817	50.7%	48.5%
5	68,285	71,550	5,889	8.6%	8.2%	409	0.6%	0.6%	85	0.1%	0.1%	6,383	9.4%	8.9%
6	95,098	98,681	21,442	22.6%	21.7%	7,185	7.6%	7.3%	2,912	3.1%	3.0%	31,540	33.2%	32.0%
7	60,909	60,909	6,719	11.0%	11.0%	1,000	1.6%	1.6%	261	0.4%	0.4%	7,980	13.1%	13.1%
8	130,311	130,311	1,365	1.1%	1.1%	90	0.1%	0.1%	367	0.3%	0.3%	1,821	1.4%	1.4%
9	54,228	56,492	2,088	3.9%	3.7%	219	0.4%	0.4%	186	0.3%	0.3%	2,493	4.6%	4.4%
10	68,189	69,524	4,295	6.3%	6.2%	1,592	2.3%	2.3%	1,044	1.5%	1.5%	6,931	10.2%	10.0%
Nation	**627,679**	**650,043**	**88,779**	**14.1%**	**13.7%**	**17,436**	**2.8%**	**2.7%**	**6,214**	**1.0%**	**1.0%**	**112,429**	**17.9%**	**17.3%**

[1] Reach miles include both freshwater and estuarine RF1 reaches.

Table 10-9: Estimated Water Quality Improvements Under Option 3[1]

EPA Region	Baseline Scenario		Water Quality Improvements by WQI Change											
			0.01 < ΔWQI < 0.1			0.1 < ΔWQI < 0.5			0.5 < ΔWQI			Total Improved Reaches		
	Reach Miles Modeled	Total Reach Miles in RF1 Network	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles
1	16,182	18,324	3,717	23.0%	20.3%	301	1.9%	1.6%	41	0.3%	0.2%	4,059	25.1%	22.2%
2	15,140	16,110	2,142	14.2%	13.3%	5	<0.1%	<0.1%	0	0.0%	0.0%	2,147	14.2%	13.3%
3	28,904	33,617	6,084	21.1%	18.1%	163	0.6%	0.5%	30	0.1%	0.1%	6,277	21.7%	18.7%
4	90,435	94,525	40,092	44.3%	42.4%	8,244	9.1%	8.7%	1,610	1.8%	1.7%	49,945	55.2%	52.8%
5	68,285	71,550	7,228	10.6%	10.1%	630	0.9%	0.9%	85	0.1%	0.1%	7,943	11.6%	11.1%
6	95,098	98,681	23,679	24.9%	24.0%	8,228	8.7%	8.3%	3,487	3.7%	3.5%	35,395	37.2%	35.9%
7	60,909	60,909	7,473	12.3%	12.3%	1,300	2.1%	2.1%	319	0.5%	0.5%	9,092	14.9%	14.9%
8	130,311	130,311	3,677	2.8%	2.8%	111	0.1%	0.1%	376	0.3%	0.3%	4,164	3.2%	3.2%
9	54,228	56,492	2,834	5.2%	5.0%	282	0.5%	0.5%	222	0.4%	0.4%	3,338	6.2%	5.9%
10	68,189	69,524	4,405	6.5%	6.3%	1,812	2.7%	2.6%	1,170	1.7%	1.7%	7,387	10.8%	10.6%
Nation	**627,679**	**650,043**	**101,332**	**16.1%**	**15.6%**	**21,075**	**3.4%**	**3.2%**	**7,340**	**1.2%**	**1.1%**	**129,747**	**20.7%**	**20.0%**

[1] Reach miles include both freshwater and estuarine RF1 reaches.

Table 10-10: Estimated Water Quality Improvements Under Option 4[1]

EPA Region	Baseline Scenario: Reach Miles Modeled	Baseline Scenario: Total Reach Miles in RF1 Network	0.01 < ΔWQI < 0.1: Reach Miles	0.01 < ΔWQI < 0.1: % of Reach Miles Modeled	0.01 < ΔWQI < 0.1: % of Total Reach Miles	0.1 < ΔWQI < 0.5: Reach Miles	0.1 < ΔWQI < 0.5: % of Reach Miles Modeled	0.1 < ΔWQI < 0.5: % of Total Reach Miles	0.5 < ΔWQI: Reach Miles	0.5 < ΔWQI: % of Reach Miles Modeled	0.5 < ΔWQI: % of Total Reach Miles	Total Improved Reaches: Reach Miles	Total Improved Reaches: % of Reach Miles Modeled	Total Improved Reaches: % of Total Reach Miles
1	16,182	18,324	2,338	14.5%	12.8%	156	1.0%	0.9%	29	0.2%	0.2%	2,522	15.6%	13.8%
2	15,140	16,110	1,433	9.5%	8.9%	5	<0.1%	<0.1%	0	0.0%	0.0%	1,438	9.5%	8.9%
3	28,904	33,617	4,620	16.0%	13.7%	108	0.4%	0.3%	30	0.1%	0.1%	4,758	16.5%	14.2%
4	90,435	94,525	38,507	42.6%	40.7%	7,159	7.9%	7.6%	1,465	1.6%	1.6%	47,130	52.1%	49.9%
5	68,285	71,550	5,979	8.8%	8.4%	377	0.6%	0.5%	85	0.1%	0.1%	6,441	9.4%	9.0%
6	95,098	98,681	21,693	22.8%	22.0%	7,819	8.2%	7.9%	3,313	3.5%	3.4%	32,825	34.5%	33.3%
7	60,909	60,909	6,860	11.3%	11.3%	1,205	2.0%	2.0%	280	0.5%	0.5%	8,345	13.7%	13.7%
8	130,311	130,311	1,282	1.0%	1.0%	90	0.1%	0.1%	367	0.3%	0.3%	1,739	1.3%	1.3%
9	54,228	56,492	2,166	4.0%	3.8%	197	0.4%	0.4%	186	0.3%	0.3%	2,548	4.7%	4.5%
10	68,189	69,524	3,894	5.7%	5.6%	1,380	2.0%	2.0%	943	1.4%	1.4%	6,217	9.1%	8.9%
Nation	**627,679**	**650,043**	**88,772**	**14.1%**	**13.7%**	**18,495**	**3.0%**	**2.9%**	**6,696**	**1.1%**	**1.0%**	**113,963**	**18.2%**	**17.5%**

[1] Reach miles include both freshwater and estuarine RF1 reaches.

10.2 Willingness to Pay for Water Quality Improvements

To estimate nonmarket benefits of water quality improvements resulting from the regulation, EPA used a benefits transfer function based on meta-analysis results presented in *Appendix G* of this report. The general approach follows standard methods illustrated by Johnston et al. (2005) and Shrestha et al. (2007), among many others (see Rosenberger and Phipps 2007). This function allows the Agency to forecast willingness to pay (WTP) based on assigned values for model variables, chosen to represent a resource change in the regulation's policy context. EPA's meta-analysis results imply a simple benefit function of the following general form:

$$\ln(\text{WTP}) = \text{intercept} + \sum (\text{coefficient}_i)(\text{Independent Variable Values}_i) \qquad \text{(Eq. 10-2)}$$

Here, ln*(WTP)* is the dependent variable in the meta-analysis—the natural log of WTP for water quality improvements. The metadata include independent variables characterizing specific details of the resource(s) valued, such as the baseline resource conditions; the extent of resource improvements and whether they occur in estuarine or freshwater; the geographic region and scale of resource improvements (e.g., the number of waterbodies); resource characteristics (e.g., baseline conditions, the extent of water quality change, and ecological services affected by resource improvements); characteristics of surveyed populations (e.g., users, nonusers); and other specific details of each study. *Table 10-11* provides the estimated regression equation *intercept* (5.71), variable coefficients ($coefficient_i$), and the corresponding independent variable names. *Appendix G* provides detail on the metadata, model specification, and justification for the functional form.

EPA assigned a value to each model variable corresponding with theory, characteristics of the water resources, and sites affected by the regulation and the policy context. *Table 10-11* presents a complete list of assigned variable values.

EPA followed Johnston et al. (2006) in assigning values for methodological attributes (i.e., variables characterizing the study methodology used in the original source studies), which are set at mean values from the metadata except in cases where theoretical considerations dictate alternative specifications. This follows general guidance from Bergstrom and Taylor (2006) that meta-analysis benefit transfer should incorporate theoretical expectations and structures, at least in a weak form. In this instance, three of the methodology variables, *discrete*, *WQI_study*, and *outlier_bids* are all included with an assigned value of one. *Year_index* is given the value of 9.68, which corresponds to the mean year that the studies were conducted, 2002. *Nonparam* is set to zero because most studies included in metadata used parametric methods to estimate WTP values. Other study and methodology variables (*volunt, mail, lump_sum, non_reviewed, median_WTP*) are assigned a zero value.

EPA used state-specific median household income, as provided by the U.S. Census Bureau's 2006 American Communities Survey (U.S. Census Bureau 2006a), to assign a value for the income variable *(income).* To be consistent with the WTP estimates, which are all estimated in 2008 dollars, EPA used the Consumer Price Index to adjust the value from the 2006 survey to 2008 dollars. The variable *nonusers* was set to zero because water quality improvements resulting from reduced sediment discharge from construction sites would benefit both users and nonusers of the affected resources.

The regulation is expected to affect water quality at a regional level because construction sites are located throughout the entire United States. The Agency set a dummy variable denoting multiple regions (*mr*) to zero because the expected magnitude of water quality improvement is relatively modest and, as a result, EPA's analysis focuses primarily on in-state or local resource improvements. The Mountain Plain regional dummy variable (*mp*) is set to zero because the magnitude of the regional effect suggests that spurious or otherwise unexplained effects (e.g., the effect of specific researchers who appear more than once in the data) may drive their overall magnitude. *Appendix G* provides more detail on regression results.

To account for the regional scale of the water quality effect in freshwater bodies resulting from the regulation, the variable *regional_fresh* is assigned a value of one. Other variables relating to waterbody type (i.e., *single_lake, single_river, salt_pond, multiple_river, num_riv_pond*) are set to zero. In estimating WTP for water quality improvements in estuaries and coastal reaches, EPA set all waterbody type variables (including *regional_fresh*) to zero because the default value is a resource change taking place in estuaries.[19]

Water quality improvements resulting from the regulation are likely to enhance a variety of water resource uses, including fishing, swimming, and boating. Therefore, variables denoting multiple uses (*allmult*) and recreational fishing *(fish_use)* are assigned a value of one, while the variable denoting nonspecified uses (*nonspec)* is set to zero. However, the variable *fishplus* is given a value of zero because it is unlikely that the regulation will cause more than a 50 percent increase in the fish population. Baseline water quality (*baseline*) and change in water quality (*quality_ch*) are assigned state-specific values as described in *Section 10.1*. For a broader discussion of issues involved in the specification of variable levels for meta-analysis benefit transfer, see Johnston et al. (2005, 2006), among others.

[19] The water quality model used in this analysis (SPARROW) does not go beyond the terminal reach to estimated water quality in large waterbodies, and these waterbodies are not fully characterized. However, EPA did calculate change in WQI for several more complex systems such as the Great Lakes and estuaries using ambient water quality data from other sources discussed in *Section 11.1.4.* EPA used mixing zone dilution factors to estimate the expected change in WQI in complex waterbodies under the post-compliance scenario (see *Chapter 6* for details).

Table 10-11: Independent Variable Assignments				
Variable Type	**Variable**	**Coefficient**	**Assigned Value**	**Explanation**
Study and Methodology Variables	*intercept*	5.7109		.
	year_index	-0.08043	9.68	Set to 9.68 to reflect the mean year that the studies in the dataset were conducted.
	discrete	-0.1248	1	Set to *one* to reflect survey efforts that employed discrete choice elicitation methods, which are preferred over other approaches, such as open-ended and payment card methods.
	volunt	-1.3233	0	Set to *zero* because hypothetical voluntary payment mechanisms are not even potentially incentive compatible (Mitchell and Carson 1989).
	mail	-0.2013	0	Set to *zero* because mail surveys are believed to be of less quality than in-person interviews.
	lump_sum	0.5569	0	Set to *zero* because the policy option will be paid for over a period of years.
	WQI	-0.3275	1	Set to *one* because of the methodological use of the WQI in the meta-analysis.
	nonparam	-0.6698	0	Set to *zero* because most studies used in the meta analysis used regression analysis to calculate willingness to pay values.
	non_reviewed	-0.2718	0	Set to *zero* because studies published in peer-reviewed journals are preferred.
	median_WTP	-0.5358	0	Set to *zero* because only average or mean WTP values in combination with the number of affected households will mathematically yield total benefits if the distribution of WTP is not perfectly symmetrical.
	outlier_bids	-0.8837	1	Set to *one* because survey data that exclude such responses are preferable; outlier bids are often excluded from the analysis of stated preference data because these bids (often identified as greater than a certain percentage of a respondent's income) may indicate that a respondent did not consider his or her budget constraints and or supplementary goods.
Surveyed Population	*income*	0.0000027	Varies	Median annual household income data from the 2006 American Communities Survey; median household income values assigned separately for each state (U.S. Census Bureau 2006a).
	nonusers	-0.4036	0	Set to *zero* in order to estimate the total value for aquatic habitat improvements, including both use and nonuse values; for nonuser population, the total value of water resource improvements includes nonuse values only (Freeman 2003).

Table 10-11: Independent Variable Assignments				
Variable Type	**Variable**	**Coefficient**	**Assigned Value**	**Explanation**
Waterbody Type Variables	*single_river*	-0.4279	0	For valuing improvement in freshwater bodies, *regional_fresh* is set to *one* because the expectation is that multiple freshwater bodies within a state would be affected by the regulation. For saltwater reaches, all waterbody type variables are set to zero because the default value in the model is a resource change taking place in estuaries.
	single_lake	-0.06316	0	
	multiple_river	-1.4752	0	
	regional_fresh	0.1588	Varies	
	salt_pond	0.9849	0	
Geographic Region and Scale Variables	*num_riv_ pond*	0.1173	0	Indicates the number of rivers or salt ponds affected by a policy, and is set to *zero* because this analysis assumes that the effluent guidelines will affect the entire watershed/region; this variable assignment is constant across study regions.
	mr	-0.8846	0	Regional variables are omitted from the predictive portion of the analysis (i.e., set to *zero*) because the regression results suggest that these variables may be picking spurious or other unexplained effects (e.g., author's effect).
	mp	1.6337	0	
Resource Improvement Variables	*allmult*	-0.3728	1	Set to *one* because it is assumed that multiple species would benefit from water quality improvements
	nonspec	-0.4042	0	Set to *zero* because it is assumed that multiple species would benefit from water quality improvements.
	lnquality_ch	0.4065	Varies	Set to the natural log of average change in the WQI for a given analytic scenario.
	fish_use	-0.3317	1	Set to *one* because the regulation is expected to benefit a variety of aquatic species and therefore enhance recreational fishing opportunities.
	fishplus	0.4432	0	Set to *zero* because the regulation is not expected to result in a fish population change of 50 percent or greater.
	lnbase	0.02610	Varies	Set to the natural log of the average base water quality for a given state.

Economic values of water quality improvements are calculated at the state level and organized by analytic scenario. EPA calculated the state-level average of baseline and post-compliance WQI changes corresponding to 12 analytic scenarios reflecting combinations of four levels of baseline water quality conditions (WQI≤ 25, 25<WQI≤50, 50<WQI≤70, and 70<WQI) and three different ranges of the expected water quality improvements (0.01< ΔWQI≤0.1, 0.1<ΔWQI≤0.5, 0.5<ΔWQI). EPA then calculated state-level WTP values for each scenario.

For each analytic scenario, coefficient estimates for each variable, taken from meta-analysis results *(Table 10-11,* column 3) are multiplied by the variable levels chosen above *(Table 10-11,* column 4). The sum of these products represents the predicted natural log of WTP (*ln_WTP*) for a given analytic scenario, as indicated by Equation 10-2. The final step uses a standard formula to transform this predicted natural log into the desired WTP estimate. This formula is given by:

$$WTP = \exp(ln\_WTP + \sigma_e^2/2) \quad \text{(Eq. 10-3)}$$

Where:

$\exp(\cdot)$ = the exponential operator

ln_WTP = the predicted natural log of WTP for a given analytic scenario

σ_e^2 = the model residual variance (0.1876) taken from *Table G-3* in *Appendix G*.

The total WTP regression model presented above can be used to predict WTP for each of the studies in the database; however, estimates derived from regression models are subject to some degree of error and uncertainty. To better characterize the uncertainty or error bounds around predicted WTP, EPA used a procedure described by Krinsky and Robb (1986). The procedure involves sampling the variance–covariance matrix of the estimated coefficients, which is a standard output of the statistical package used to estimate the meta-analysis model. WTP values are then calculated for each drawing from the variance–covariance matrix and an empirical distribution of WTP values is constructed. By varying the number of drawings, it is possible to generate an empirical distribution with a desired degree of accuracy (Krinsky and Robb 1986). The low and high estimate of WTP values is then identified based on the 10th and 90th percentile of WTP values from the empirical distribution. These bounds may help decision-makers understand the uncertainty associated with the benefit results.

EPA used the Krinsky and Robb (1986) procedure to estimate the 10^{th}, 50^{th}, and 90^{th} percentiles of total WTP for each EPA region, based on the results of the total WTP regression model. *Table 10-12* presents the results of these calculations. The Agency notes that this analysis provides confidence limits for WTP estimates related to the covariance matrix of meta-analysis parameter estimates. It does not, however, assess the sensitivity of results to changes in meta-regression model assumptions or specifications (cf. Johnston et al. 2005, 2006) or assumptions implied in benefit aggregation (cf. Loomis 1996; Loomis et al. 2000; Bateman et al. 2006). As noted above, however, the Agency has to the extent practicable made assumptions and specifications that lead to conservative benefit estimates.

Table 10-12 presents 10^{th}, 50^{th}, and 90^{th} percentiles of household WTP for water quality improvements resulting form reduced sediment discharge from construction sites by EPA region and policy option.

Table 10-12: Estimates of Annual Household Willingness to Pay for Water Quality Improvement by Region and Policy Option (2008$)

EPA Region	Option 1			Option 2			Option 3			Option 4		
	Low 10%	Mid 50%	High 90%	Low 10%	Mid 50%	High 90%	Low 10%	Mid 50%	High 90%	Low 10%	Mid 50%	High, 90%
1	$0.14	$0.87	$1.96	$0.36	$2.13	$4.74	$0.50	$2.94	$6.54	$0.30	$1.82	$4.08
2	$0.04	$0.30	$0.66	$0.17	$1.10	$2.45	$0.25	$1.63	$3.66	$0.16	$1.08	$2.42
3	$0.09	$0.56	$1.24	$0.28	$1.71	$3.79	$0.40	$2.40	$5.31	$0.29	$1.76	$3.90
4	$1.14	$4.39	$9.07	$1.90	$7.05	$14.37	$2.17	$7.87	$15.96	$1.96	$7.23	$14.73
5	$0.09	$0.49	$1.07	$0.21	$1.13	$2.45	$0.27	$1.43	$3.09	$0.22	$1.16	$2.51
6	$1.19	$3.98	$7.87	$1.88	$5.83	$11.28	$2.16	$6.59	$12.68	$2.05	$6.23	$11.98
7	$0.42	$1.62	$3.32	$0.70	$2.63	$5.34	$0.82	$2.97	$5.98	$0.76	$2.80	$5.64
8	$0.45	$0.80	$1.25	$0.47	$0.93	$1.52	$0.54	$1.29	$2.29	$0.47	$0.91	$1.48
9	$0.17	$0.64	$1.30	$0.26	$0.98	$2.01	$0.30	$1.16	$2.37	$0.26	$0.99	$2.02
10	$0.64	$2.20	$4.41	$0.98	$3.04	$5.93	$1.10	$3.33	$6.42	$0.84	$2.68	$5.26
National	**$0.49**	**$1.85**	**$3.78**	**$0.83**	**$3.10**	**$6.33**	**$0.97**	**$3.63**	**$7.41**	**$0.86**	**$3.17**	**$6.45**

As shown in *Table 10-12* the mid-point estimates of household WTP for water quality improvements resulting from the regulation range from $1.85 to $3.63 per household per year under different policy options. The estimated WTP values vary greatly across EPA regions depending on the expected water quality improvements and water resource and population characteristics in a given region.

Option 1 is estimated to improve ambient water quality in 11.6 percent of RF1 reach miles modeled in the analysis. This option results in the least water quality improvements across all EPA regions. EPA Region 4 is expected to the have the greatest improvements in water quality under this option. The estimated annual WTP per household in Region 4 is $4.39 (mid-point estimate). Region 2 has the smallest household WTP, with a mid-point estimate of $0.30 per household. Nationwide, the estimated annual household WTP is $1.85 (mid-point estimate).

Option 2 is estimated to improve ambient water quality in 17.9 percent of RF1 reach miles modeled. The estimated national average WTP for water quality improvements resulting from the regulation is $3.10 per household per year. Regions 4 and 6 are estimated to see improvements in water quality in 50.7 and 33.2 percent of their total RF1 reach miles (*Table 10-8*). EPA's analysis suggests that Region 4 households would be willing to pay the most, with a mid-point estimate of $7.05 per household per year, for water quality improvements resulting from the regulation. Region 6 has the second largest household WTP of $5.83. Conversely, households located in Region 2 are estimated to have the lowest WTP for water quality improvements from the regulation, $1.10.

Policy Option 3 yields the most significant results overall in terms of reach miles expected to improve under the post-compliance scenario (20.7 percent). The estimated scale of improvements ranges from 3.2 percent to 55.6 percent of total reach miles in Region 8 and 4, respectively (*Table 10-9*). Nationwide, the estimated annual per-household WTP for water quality improvements under Option 3 is $3.63 (mid-point estimate).

Option 4 is estimated to generate improvements in 18.2 percent of reach miles nationally. These improvements result in a national household WTP of $3.17 per year (mid-point estimate).. As with the other options, Region 4 has the largest annual household WTP value with a mid-point estimate of $7.23. Conversely, EPA Region 8 has the lowest WTP value with a mid-point estimate of $0.91.

10.3 Estimating Total WTP for Water Quality Improvements

For each policy option, EPA calculated reach-level WTP as follows. First EPA estimated mean state-level per-household WTP for each combination of the baseline water quality category ($WQI_{baseline}$) and the expected change in WQI (ΔWQI). Then, the Agency assigned each reach in the analysis a mean household WTP value based on reach location, baseline water quality, and change in water quality. The WTP was then multiplied by the number of households in a given state in 2006 and the percentage of reach miles in that state that comprise a given reach. The number of households per state was calculated by taking U.S. Census Bureau population estimates for 2006 for each state and dividing by the average number of people per household for a given state as reported in U.S. Census Bureau (2006a, 2006b). The total WTP equation for each reach is provided below (Equation 10-4):

$$TWTP_{reach} = WTP(WQI_{baseline}, \Delta WQI) \times StateHH \times PercentRiverMiles) \qquad \text{(Eq. 10-4)}$$

Where:

$TWTP_{reach}$ = the reach-level welfare change from improved water quality

WTP	=	the estimated state-level per-household WTP for water quality improvement for a given combination of the baseline water quality category ($WQI_{baseline}$) and the expected change in water quality under the post-compliance scenario (ΔWQI)
StateHH	=	the number of households in a given state
PercentRiverMiles	=	the percentage of total reach miles in a given state that are comprised of a given reach.

Finally, EPA aggregated reach-level benefits to the regional level. The regional benefits for the ten EPA regions were then combined to calculate the national benefit of the regulation. *Table 10-13* presents estimated benefits of the regulation by EPA region and policy option.

Table 10-13 :Regional Willingness to Pay for Water Quality Improvement (Millions 2008$)

EPA Region	Option 1			Option 2			Option 3			Option 4		
	Lower 10% Bound	Mean	Upper 90% Bound	Lower 10% Bound	Mean	Upper 90% Bound	Lower 10% Bound	Mean	Upper 90% Bound	Lower 10% Bound	Mean	Upper 90% Bound
1	$0.81	$4.93	$11.10	$2.06	$12.07	$26.84	$2.84	$16.67	$37.03	$1.71	$10.32	$23.09
2	$0.45	$3.10	$6.94	$1.76	$11.50	$25.67	$2.63	$17.14	$38.36	$1.71	$11.30	$25.35
3	$1.07	$6.45	$14.23	$3.27	$19.70	$43.59	$4.58	$27.61	$61.08	$3.36	$20.30	$44.94
4	$26.41	$101.44	$209.34	$43.94	$162.70	$331.84	$50.10	$181.69	$368.63	$45.36	$167.04	$340.13
5	$1.78	$9.92	$21.64	$4.27	$22.97	$49.85	$5.43	$29.01	$62.88	$4.37	$23.52	$51.02
6	$15.62	$52.44	$103.61	$24.72	$76.74	$148.49	$28.45	$86.74	$166.91	$27.01	$82.05	$157.72
7	$2.28	$8.86	$18.12	$3.84	$14.37	$29.15	$4.49	$16.24	$32.67	$4.17	$15.27	$30.81
8	$1.76	$3.17	$4.91	$1.85	$3.66	$5.97	$2.12	$5.08	$9.03	$1.84	$3.60	$5.83
9	$2.67	$9.95	$20.26	$4.04	$15.32	$31.28	$4.72	$18.09	$36.97	$4.08	$15.43	$31.47
10	$2.90	$10.01	$20.09	$4.45	$13.86	$27.03	$5.03	$15.16	$29.25	$3.82	$12.21	$23.97
National	**$55.75**	**$210.27**	**$430.24**	**$94.19**	**$352.90**	**$719.72**	**$110.39**	**$413.41**	**$842.81**	**$97.44**	**$361.04**	**$734.34**

EPA estimates that total annual benefits of water quality improvements resulting from reduced sediment discharge from construction sites range from $210.3 million under Option 1 to $413.4 million under Option 3. The estimated mean regional benefits vary from $3.1 to $181.7 million per year, depending on the level of construction activity, average rainfall, water resource characteristics, and resident population in a given region and stringency of the policy option.

As shown in *Table 10-13,* Option 1 generates the least water quality improvements of the four regulatory options. Thus, this option yields the smallest benefits at the regional and national levels. The national benefits of water quality improvements under this option range from $55.8 million to $430.2 million per year, with a mid-point estimate of $210.3 million. Region 4 gains the most benefit from water quality improvement, with a total value of $101.4 million per year (48.2 percent of the total national benefits). Region 6 has the second largest benefits ($52.4 million per year), which account for 24.9 percent of the total national benefits.

Under Option 2, the mid-point estimate of national benefits is $352.9 million per year, with low and high estimates of $94.2 million and $719.7 million per year. Region 4 gains the most from water quality improvements resulting from the regulation ($162.7 million per year). EPA Region 8 receives the smallest annual benefits, $3.7 million

Under Option 3, the estimated national benefits of water quality improvement from the regulation are $413.4 million per year, with a low estimate of $110.4 and a high estimate of $842.8 million. As with the other policy options Region 4 receives the largest benefits from water quality improvements, accounting for 43.9 percent ($181.7 million per year) of the total national benefits. Region 8 is estimated to gain least under Option 3, with the total regional benefits estimated at $5.1 million per year.

Option 4 is expected to generate national benefits of $361.0 million per year with a low estimate of $97.4 million and a high estimate of $734.3 million. EPA Region 4 gains the largest benefits, with a total of $159.9 million per year. Region 8 has the smallest improvements in water quality under this option, with improvements in only 1.3 percent of reach miles, and thus has the smallest annual benefits ($3.6 million).

10.4 Uncertainty and Limitations

A number of issues are common to all benefit transfers. Benefit transfer involves adapting research conducted for another purpose in the available literature to address the policy questions at hand. Because benefits analysis of environmental regulations rarely affords enough time to develop original stated preference surveys that are specific to the policy effects, benefit transfer is often the only option to inform a policy decision. As a result, they are nearly universal in benefit-cost analyses (Smith et al. 2002).

Benefit transfers are by definition characterized by a difference between the context in which resource values are estimated and that in which benefit estimates are desired (Rosenberger and Phipps 2007). The ability of meta-analysis to adjust for the influence of study, economic, and resource characteristics on WTP can minimize, but not eliminate, potential biases (Smith et al. 2002; Rosenberger and Stanley 2006; Rosenberger and Phipps 2007). As is typical in applied benefit transfers, the meta-analysis model used in this analysis provides a close, but not perfect, match to the context in which values are desired. Therefore, some beneficial effects associated with reducing sediment discharges from construction sites are not accounted for in the estimated WTP values. For example, surface water valuation studies included in meta-data focused primarily on changes in ambient water quality. Other effects associated with sediment discharges (e.g., sedimentation of river beds, stability and erosion of river banks, and changes in the stream carrying capacity) were not included in the original study scenarios and, as a result, from the estimated WTP for reducing sediment discharges from construction sites.

Some related and additional limitations inherent to the meta-analysis model and the subsequent benefit transfer include:

- The Agency notes, as detailed by Loomis (1996), Loomis et al. (2000) and Bateman et al. (2006), among others, that there are numerous uncertainties and associated assumptions required to aggregate WTP across spatial jurisdictions. While these uncertainties are well known, the literature does not agree on appropriate, standardized guidance for benefit aggregations, and applied benefit-cost analysis almost universally requires simplifying assumptions in order to generate defensible welfare aggregations. In an ideal context, analysts would have information necessary to estimate spatially referenced distance decay relationships for all changes resulting from policies under consideration (cf. Bateman et al. 2006). However, the Agency notes that even the most advanced literature provides only simple illustrations of such issues, and none methodologically sufficient to support regulatory analysis. In analyzing benefits of the final regulation, EPA assumed that households would gain no benefits from water quality improvements in aquatic resources located outside of their state of residence. As a result, the population considered in the benefits analysis of the regulation does not represent all the households that are likely to hold values for water resources in a given state. Residents of other states may hold values for water resources outside of their home state, in particular if such resources have personal, regional, or national significance. Even if per household WTP for out-of-state residents are small they can be very large in the aggregate if these values are held by a substantial fraction of the population
- Some resource valuation studies have found that respondents in the typical contingent market situation may overstate their WTP compared to their likely behavior in a real-world situation. However, the magnitude of hypothetical bias on the estimated WTP is uncertain. Following standard benefit transfer approaches, including meta-analytic transfers, this analysis proceeds under the assumption that each source study provides a valid, unbiased estimate of the welfare measure under consideration (cf. Moeltner et al. 2007; Rosenberger and Phipps 2007). To minimize potential hypothetical bias, EPA set independent variable values to reflect best benefit transfer practices.
- The estimation of WTP may be sensitive to differences in the environmental water quality measures. Studies that did not use the WQI were mapped to the WQI so a comparison could be made across studies. The dummy variable *(WQI)* captures the effect of a study using *(WQI*=1) or not using the WQI *(WQI*=0). It was found that studies that did not use the WQI had lower WTP values. This may indicate that there may have been some systematic biases in the mapping of studies that did not use the WQI. In analyzing the benefits of this regulation, EPA set the WQI to one to reduce uncertainty in WTP estimates associated with studies that did not include WQI as a native survey instrument. See *Appendix G* for a detailed discussion of water quality measures used in the original studies included in the meta-analysis.
- Transfer error may occur when benefit estimates from a study site are adopted to forecast the benefits of a policy site. Rosenberger and Stanley (2006) define transfer error as the difference between the transferred and actual, generally unknown, value. While meta-analysis is fairly accurate when estimating benefit function, transfer error may be a problem in cases where the sample size is small. While meta-analyses have been shown to outperform other

function-based transfer methods in many cases, this result is not universal (Shrestha et al. 2007). This notwithstanding, results reviewed by Rosenberger and Phipps (2007) are "very promising" for the performance of meta-analytic benefit transfers relative to alternative transfer methods.

Additional limitations and uncertainties are associated with the use of WQI to link water quality changes from reduced sediment discharges to effects on human uses and support for aquatic and terrestrial species habitat:

- The estimated changes in WQI reflect only water quality improvements resulting directly from reductions in total suspended solid and nutrient loadings. They do not include improvements in water quality indicators indirectly associated with pollutant loadings, (e.g., dissolved oxygen). This is likely to result in underestimation of the expected water quality changes resulting from the proposed regulation because the combined impact of several pollutants on ambient water quality is likely to be greater than the sum of the individual impacts of reducing concentrations of sediments and nutrients.
- The WQI index used in EPA's analysis uses TSS concentrations as a proxy for water turbidity. It does not include turbidity as a separate parameter in calculation of the WQI. This omission may understate the expected change in WQI resulting from the final regulation.
- The methodology used to translate in-stream sediment and nutrient concentrations into sub-index scores employs nonlinear transformation curves. Water quality changes that fall outside of the sensitive part of the transformation curve (i.e., above/below the upper/lower bounds) yield no benefit in the analysis.

Limitations and uncertainty associated with the use of SPARROW are discussed in *Chapter 6* of this report.

11 Total Estimated Benefits

This chapter summarizes findings from EPA's analysis of the expected benefits from the regulation. EPA considered a wide range of policy options in developing this regulation and presents here the three policy options for which a full cost-benefit analysis was performed. These options are:

- Option 1: nonnumeric effluent limitations on all sites.
- Option 2: in addition to the requirements of Option 1, active sediment treatment is required on sites with 30 or more acres disturbed at one time and treated discharge is subject to a turbidity standard of 13 NTU.
- Option 3: in addition to the requirements of Option 1, active sediment treatment is required on sites with 10 or more acres disturbed at one time and treated discharge is subject to a turbidity standard of 13 NTU.
- Option 4: in addition to the requirements of Option 1, passive sediment treatment is required on sites with 10 or more acres disturbed at one time, and treated discharge is subject to a turbidity standard of 280 NTU.

Chapter 1 describes these policy options in more detail.

11.1 Summary of the Estimated Benefits

The Agency analyzed benefits of the regulation in four benefits categories: navigation, water storage, drinking water treatment, and nonmarket benefits of water quality improvement. The previous chapters of this report provide the details of the methods and data used in the analysis of the four monetized benefit categories (i.e., list benefit categories). See *Chapters 7, 8,* and 9 for a discussion of the avoided cost methods used to estimate benefits to navigation, water storage, and treatment. *Chapter 10* and *Appendix G* provide a discussion of the methods used to estimate the willingness to pay (WTP) for water quality improvements resulting from the policy options considered for the regulation.

EPA was unable to quantify benefits stemming from:

- Reduced damages to industrial and agricultural water users
- Improved market value of properties near surface waters
- Reduced cost of stormwater system maintenance and flood damages
- Benefits to commercial fishing and shellfishing
- Reduced cost of drinking water treatment to improve taste and odor resulting from eutrophication of drinking water sources.

Chapter 5 of this report describes these benefits qualitatively. In addition, limitations inherent in EPA's analysis of water quality improvements and monetization of benefits are likely to result in understatement of benefits resulting from the final regulation. *Section 11.2* below summarizes key limitations of EPA's analysis of benefits stemming from reducing construction site discharges..

Table 11-1 presents low, midpoint, and high estimates of benefits under each policy option, consisting of benefits to navigation, water storage, drinking water treatment, and WTP. *Table 11-2, Table 11-3, Table*

*11-4,*and *Table 11-5* detail total benefits by EPA region, policy option, and estimate range. It should be noted that these tables incorporate the confidence intervals of 10, 50, and 90 percent from the WTP analysis into the low, mid, and high sensitivity analyses performed for the avoided cost estimates. Though these are conceptually different, they are both intended to present a range of values to account for some of the uncertainty inherent in these estimates. The sensitivity analyses create a range by varying EPA's assumptions underlying the analysis, while the confidence interval presents high and low bounds from the meta-analysis regression.

All tables present benefits for navigable waterway and reservoir dredging calculated using both 3 and 7 percent discount rates. Because the discount rate only applies to two of the four monetized benefits categories, which represent at most 5 percent of total benefits, varying it has little effect on the total benefits estimate. EPA calculated benefits for drinking water treatment and WTP using a single-year timeframe, which did not require discounting or additional calculations to present annual values. All benefits presented reflect annual values. The remaining discussion presents the benefits estimates assuming a 3 percent discount rate; the associated tables present results for both discount rates.

Total national benefits vary significantly among the three regulatory options. Under Option 1, the estimated benefits range from approximately $59.0 million to approximately $434.2 million, with a midpoint estimate of $214.0 million. Estimated avoided costs range from $3.3 million to $4.1 million, with a midpoint of $3.8 million, and WTP ranges from $55.8 to $430.2 million , with a midpoint estimate of $210.3 million.

For Option 2, the estimated benefits range from $100.5 million to $727.4 million, with a midpoint estimate of $360.1 million. The estimated WTP for water quality improvements from reduced sediment discharges from construction sites under Option 2 ranges from $94.2 to $719.7 million, with a midpoint value of $352.9 million. Estimated cost savings range from $6.3 million to $7.7 million per year, with a midpoint estimate of $7.2 million.

Under Option 3, total benefits are estimated to be between $118.0 and $852.2 million, with a midpoint estimate of $422.3 million. The avoided costs are estimated to be between $7.7 and $9.4 million per year, with a midpoint estimate of $8.9 million. WTP under Option 3 ranges from $110.4 million to $842.8 million , with a midpoint estimate of $413.4 million.

Under Option 4, the final regulation, the estimated benefits range from $104.3 million to $742.7 million, with a midpoint estimate of $368.9 million. Nonmarket benefits estimated based on household WTP for surface water quality improvements account for 93, 98, and 99 percent of total benefits from the regulation in the low, mid, and high estimates, respectively. The estimated WTP for water quality improvements from reduced sediment discharges from construction sites under Option 4 ranges from $97.4 to $734.3 million, with a midpoint estimate of $361.0 million. The estimated cost savings to industry and government through reduced costs of navigable waterway maintenance, reservoir dredging, and drinking water treatment range from $6.8 million to $8.3 million per year, with a midpoint estimate of $7.9 million. Under Option 4, avoided cost benefits account for 7, 2, and 1 percent of total benefits in the low, mid, and high estimates, respectively. Because this option requires passive treatment at sites with more than 10 acres of land disturbed and establishes a numeric effluent limit, its benefits are more than double those of Option 1, which does not establish numeric criteria for sediment discharge. It also produces more benefits than Option 2, which requires active treatment of sediment but on fewer sites. Benefits under Option 4 are lower than those under Option 3, which would require active sediment treatment on the same sites where Option 4 requires passive treatment, which is less burdensome.

Table 11-1: Annual Total National Benefits by Benefits Category (millions of 2008$)

Benefit Category	3% Discount Rate			7% Discount Rate		
	Low	Mid	High	Low	Mid	High
Option 1						
Navigation	$1.0	$1.3	$1.3	$1.0	$1.2	$1.3
Water Storage[1]	$1.3	$1.4	$1.5	$1.1	$1.2	$1.4
Drinking Water[1]	$1.0	$1.2	$1.3	$1.0	$1.2	$1.3
Avoided Costs	*$3.3*	*$3.8*	*$4.1*	*$3.1*	*$3.7*	*$4.0*
WTP[1]	$55.8	$210.3	$430.2	$55.8	$210.3	$430.2
Total[2]	**$59.0**	**$214.1**	**$434.3**	**$58.8**	**$213.9**	**$434.2**
Option 2						
Navigation	$2.1	$2.6	$2.8	$2.1	$2.5	$2.7
Water Storage[1]	$2.7	$2.9	$3.0	$2.2	$2.6	$3.0
Drinking Water[1]	$1.4	$1.8	$1.9	$1.4	$1.8	$1.9
Avoided Costs	*$6.3*	*$7.2*	*$7.7*	*$5.8*	*$6.9*	*$7.5*
WTP[1]	$94.2	$352.9	$719.7	$94.2	$352.9	$719.7
Total[2]	**$100.5**	**$360.1**	**$727.4**	**$99.9**	**$359.8**	**$727.2**
Option 3						
Navigation	$2.7	$3.3	$3.4	$2.6	$3.2	$3.4
Water Storage[1]	$3.3	$3.6	$3.8	$2.8	$3.2	$3.7
Drinking Water[1]	$1.7	$2.1	$2.1	$1.7	$2.1	$2.1
Avoided Costs	*$7.7*	*$8.9*	*$9.4*	*$7.0*	*$8.4*	*$9.2*
WTP[1]	$110.4	$413.4	$842.8	$110.4	$413.4	$842.8
Total[2]	**$118.0**	**$422.3**	**$852.2**	**$117.4**	**$421.8**	**$852.0**
Option 4						
Navigation	$2.4	$2.9	$3.0	$2.3	$2.8	$3.0
Water Storage[1]	$3.0	$3.2	$3.4	$2.5	$2.9	$3.3
Drinking Water[1]	$1.5	$1.8	$1.9	$1.5	$1.8	$1.9
Avoided Costs	*$6.8*	*$7.9*	*$8.3*	*$6.3*	*$7.5*	*$8.2*
WTP[1]	$97.4	$361.0	$734.3	$97.4	$361.0	$734.3
Total[2]	**$104.3**	**$368.9**	**$742.7**	**$103.7**	**$368.5**	**$742.5**

[1] These savings were calculated for a one-year timeframe, did not require discounting, and are equal under both discount rates.
[2] Totals may not equal the sum of categories due to rounding.

Table 11-2, *Table 11-3*, *Table 11-4*, and *Table 11-5* detail total monetized benefits (including benefits to navigation, water storage, drinking water, and water quality) by region. Region 4 benefits the most from this regulation under all policy options, as it experiences the most widespread changes in terms of improved reach miles and the most significant reductions in sediment concentrations in these reach miles. This leads to higher WTP estimates in Region 4, which account for the largest proportion of benefits. Region 6 benefits second most, though monetized benefits in this region are about half of those in Region 4. Regions 4 and 6 together account for more than half of the benefits under all of the options. For Region 4, midpoint benefits estimates are $102.4, $164.7, $184.2, and $169.2 million, respectively under the four policy options. For Region 6, midpoint benefits estimates for the four options are $54.1, $80.1 $90.9, and $86.0 million, respectively.

Table 11-2: Annual Total National Benefits Under Option 1 (millions of 2008$)

EPA Region	Avoided Dredging Costs Discounted at 3%[1]			Avoided Dredging Costs Discounted at 7%[1]		
	Low	Mid	High	Low	Mid	High
1	$0.8	$5.0	$11.1	$0.8	$5.0	$11.1
2	$0.5	$3.2	$7.1	$0.5	$3.2	$7.1
3	$1.1	$6.5	$14.3	$1.1	$6.5	$14.3
4	$27.3	$102.4	$210.4	$27.2	$102.4	$210.4
5	$2.3	$10.6	$22.3	$2.3	$10.6	$22.3
6	$17.1	$54.1	$105.3	$16.9	$54.0	$105.3
7	$2.3	$8.9	$18.2	$2.3	$8.9	$18.2
8	$1.8	$3.2	$4.9	$1.8	$3.2	$4.9
9	$2.7	$10.0	$20.3	$2.7	$10.0	$20.3
10[2]	$3.1	$10.3	$20.4	$3.1	$10.2	$20.4
Total[3]	**$59.0**	**$214.1**	**$434.3**	**$58.8**	**$213.9**	**$434.2**

[1] Only avoided costs of dredging navigable waterways and reservoirs required discounting and annualization. Avoided costs of drinking water treatment and willingness-to-pay were estimated on a single-year basis.
[2] Benefits estimates in this region are not zero, but less than $500 annually.
[3] Totals may not be equal to the sum of regional data because the WTP model estimates the national total rather than summing regional totals.

Table 11-3: Annual Total National Benefits Under Option 2 (millions of 2008$)

EPA Region	Avoided Dredging Costs Discounted at 3%[1]			Avoided Dredging Costs Discounted at 7%[1]		
	Low	Mid	High	Low	Mid	High
1	$2.1	$12.1	$26.9	$2.1	$12.1	$26.9
2	$1.8	$11.6	$25.8	$1.8	$11.6	$25.8
3	$3.4	$19.8	$43.7	$3.4	$19.8	$43.7
4	$45.7	$164.7	$333.9	$45.5	$164.6	$333.9
5	$4.9	$23.7	$50.6	$4.9	$23.7	$50.6
6	$27.7	$80.1	$152.0	$27.4	$79.9	$152.0
7	$4.0	$14.5	$29.3	$4.0	$14.5	$29.3
8	$1.9	$3.7	$6.0	$1.9	$3.7	$6.0
9	$4.2	$15.5	$31.5	$4.2	$15.5	$31.5
10	$4.8	$14.4	$27.6	$4.8	$14.4	$27.6
Total[2]	**$100.5**	**$360.1**	**$727.4**	**$99.9**	**$359.8**	**$727.2**

[1] Only avoided costs of dredging navigable waterways and reservoirs required discounting and annualization. Avoided costs of drinking water treatment and willingness-to-pay were estimated on a single-year basis.
[2] Totals may not be equal to the sum of regional data because the WTP model estimates the national total rather than summing regional totals.

Table 11-4: Annual Total National Benefits Under Option 3 (millions of 2008$)

EPA Region	Avoided Dredging Costs Discounted at 3%[1]			Avoided Dredging Costs Discounted at 7%[1]		
	Low	Mid	High	Low	Mid	High
1	$2.9	$16.7	$37.1	$2.9	$16.7	$37.1
2	$2.7	$17.3	$38.5	$2.7	$17.3	$38.5
3	$4.7	$27.8	$61.2	$4.7	$27.8	$61.2
4	$52.3	$184.2	$371.2	$52.1	$184.0	$371.2
5	$6.1	$29.8	$63.7	$6.0	$29.7	$63.7
6	$32.2	$90.9	$171.3	$31.9	$90.7	$171.2
7	$4.7	$16.4	$32.9	$4.6	$16.4	$32.9
8	$2.1	$5.1	$9.1	$2.1	$5.1	$9.1
9	$4.9	$18.3	$37.2	$4.9	$18.3	$37.2
10	$5.5	$15.8	$30.0	$5.5	$15.8	$30.0
Total[2]	**$118.0**	**$422.3**	**$852.2**	**$117.4**	**$421.8**	**$852.0**

[1] Only avoided costs of dredging navigable waterways and reservoirs required discounting and annualization. Avoided costs of drinking water treatment and willingness-to-pay were estimated on a single-year basis.
[2] Totals may not be equal to the sum of regional data because the WTP model estimates the national total rather than summing regional totals.

Table 11-5: Annual Total National Benefits Under Option 4 (millions of 2008$)

EPA Region	Avoided Dredging Costs Discounted at 3%[1]			Avoided Dredging Costs Discounted at 7%[1]		
	Low	Mid	High	Low	Mid	High
1	$1.7	$10.4	$23.1	$1.7	$10.4	$23.1
2	$1.8	$11.4	$25.5	$1.8	$11.4	$25.5
3	$3.5	$20.4	$45.1	$3.5	$20.4	$45.1
4	$47.3	$169.2	$342.4	$47.1	$169.1	$342.4
5	$5.0	$24.2	$51.8	$5.0	$24.2	$51.8
6	$30.5	$86.0	$161.8	$30.2	$85.8	$161.8
7	$4.3	$15.4	$31.0	$4.3	$15.4	$31.0
8	$1.9	$3.6	$5.8	$1.9	$3.6	$5.8
9	$4.2	$15.6	$31.7	$4.2	$15.6	$31.7
10	$4.1	$12.6	$24.4	$4.1	$12.6	$24.4
Total[2]	**$104.3**	**$368.9**	**$742.7**	**$103.7**	**$368.5**	**$742.5**

[1] Only avoided costs of dredging navigable waterways and reservoirs required discounting and annualization. Avoided costs of drinking water treatment and willingness-to-pay were estimated on a single-year basis.
[2] Totals may not be equal to the sum of regional data because the WTP model estimates the national total rather than summing regional totals.

11.2 Sources of Uncertainty and Limitations

EPA notes that quantifying and monetizing impacts of reducing sediment discharges from construction sites is challenging. As a result, total national benefits estimates of the regulation are subject to the limitations and uncertainties inherent in the valuation approaches used for assessing benefits to navigation, water storage, drinking water treatment, and nonmarket benefits of water quality improvement. Because the combined effect of these limitations and uncertainties is likely to underestimate the national level of benefits of this regulation, the estimated benefits should be interpreted in the context of these limitations.

The preceding sections of this report discuss specific limitations and uncertainties associated with estimating benefits to navigation, water storage, drinking water treatment, and nonmarket benefits of water quality improvement. This section summarizes the limitations inherent in EPA’s analysis of reducing discharges from the construction sites.

11.2.1 Water Quality Model Limitations

To estimate benefits of reduced sediment loadings to surface water, EPA relied on SPARROW. The SPARROW model for suspended sediments has a number of limitations, some of which are inherent to the methodology and some the result of the particular model application. The key model limitations are:

- Reliance on the Reach File 1 network. While the RF1 network provides reasonably comprehensive national coverage of major rivers, streams, and other surface waterbodies, coverage is limited in certain important respects. First, RF1 network coverage is limited to the coterminous United States, thus excluding Alaska and Hawaii. In addition, while the RF1 1:500,000-scale network reaches have associated data or estimates of stream discharge and velocity that are required to specify the SPARROW model, the network excludes the majority of the nation’s total stream mileage, and smaller streams in particular. The linear coverage of the RF1 network is approximately 700,000 miles (USEPA 2007e). By contrast, coverage of the USGS National Hydrographic Dataset, at 1:24,000 - 1:100,000 scale, is currently over 7 million miles (USGS 2007b). Given that RF1 accounts only for 10 percent of the total reach miles, the impacts of construction-related sediment on smaller stream reaches are likely to be significantly understated. As construction activities may be concentrated along lower-order streams not included in the RF1 network, the relative share of total sediments contributed by construction activities may be high on these reaches during active construction phases. By contrast, the specific impacts of construction activities may diminish in importance relative to contributions from spatially extensive and diffuse land uses, including agriculture, at the level of RF1 reaches. Moreover, approximately 20–30 percent of sediment discharged from construction sites does not reach the RF1 network and is instead retained on the land surface and in smaller streams and reservoirs. These loads are omitted from the current analysis.
- Estimation of changes in nutrient concentrations associated with changes in sediment loading. The approach used to estimate changes in nutrient concentration resulting from reduction in sediment loadings is based on modeled long-term relationships between sediment and nutrient loadings within each of the modeled watersheds. This assumption follows observations from case studies discussed in *Chapter 4* of this report, which suggest a correlation between sediment loadings from construction sites and elevated nitrogen and phosphorus loadings. By using a fixed relationship, the approach assumes that nutrients are bound to the sediments and that methods used to reduce sediment runoff from construction sites are equally effective in reducing nitrogen and phosphorus runoff from these sites. There is currently insufficient information to assess the extent to which actual reductions may differ from this assumption. While phosphorus is often attached to the sediment and therefore may be more readily addressed by control measures that retain sediments on site, nitrogen is typically found in soluble forms, and sediment control measures may be less effective in reducing nitrogen loading.

- Omission of all ponds and lakes and reservoirs located off the RF1 network from the water quality analysis. All lakes, ponds, and reservoirs located off the RF1 network are not included in the SPARROW model and thus are excluded from estimation of monetized benefits. The National Water Quality Inventory Report to Congress (USEPA 2009e) reports 40.6 million acres of lakes and reservoirs in the coterminous United States. The RF1 network includes approximately 3.9 million acres or 9.5 percent of the total lakes and reservoir acres in the United States (USEPA 2007e).[20] Omission of these waterbody types from the analysis of monetized benefits is likely to lead to understatement of benefits in two benefit categories: (1) nonmarket benefits of water quality improvements resulting from the regulation and (2) reservoir dredging.
- Restriction of the water quality analysis to the description of long-term mean water quality conditions. Construction activities are, by contrast, transient in nature, extending over weeks or months. Construction activities (unlike agricultural activities) are spatially compact, so they are a sub-grid phenomenon with respect to the specification of the national scale of the SPARROW model. The restriction to mean water quality conditions precludes an analysis of the frequency with which conditions of extreme sediment transport conditions (e.g., during an active construction period) occur. Although the predicted changes in average water quality conditions may be small, the expected changes in sediment concentrations under extreme sediment transport conditions may be significant.

11.2.2 Focus on Selected Pollutants of Concern (Sediment and Nutrients)

Existing case studies of environmental impacts associated with construction activities demonstrated that a number of pollutants are found in construction site discharges, including turbidity, BOD, metals, toxic organics, trash and debris, and other miscellaneous pollutants. However, EPA's analysis of benefits from reduced construction site discharges focuses only on water quality improvements resulting directly from reductions in total suspended solid and nutrient loadings. It does not include improvements resulting from reductions in other pollutant loadings, nor does it include improvements in water quality indicators indirectly associated with pollutant loadings (e.g., increase in primary productivity such as algae growth, changes in dissolved oxygen, and turbidity). This is likely to result in underestimation of the expected water quality changes resulting from the regulation because the combined impact of several pollutants on ambient water quality conditions is likely to be greater than the sum of the individual impacts of reducing concentrations of sediments and nutrients.

11.2.3 Omission of Several Benefit Categories from the Analysis of Monetized Benefits

Due to data limitations, EPA did not estimate benefits in several benefit categories. Although the magnitude of benefits in the omitted categories is unknown, they may not be trivial. *Chapter 5* of this report provides a qualitative discussion of the omitted benefit categories. A brief summary of the omitted benefit categories is provided below:

- Market values of properties located near waterbodies. Reducing sediment discharges from construction sites is likely to increase market values of properties located in the vicinity of

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20 The estimated total lake/reservoir acres do not include the Great Lakes.

construction sites by enhancing the aesthetic quality of the affected land and water resources (e.g., reducing erosion of river banks and improving water clarity).[21]

- Flood damages. Reducing sediment discharges from construction sites is expected to reduce flooding damages by decreasing sedimentation of river beds and improving river capacity. Clark et al. (1985) estimated flooding damages attributable to sediment discharges to be $1.5 billion (2008$), annually. Therefore, even a small reduction in the frequency and severity of flooding is likely to generate significant benefits.
- Ditch maintenance. The regulation is expected to reduce the costs of ditch maintenance by reducing the amount of sediment deposited in ditches.
- Industrial water use. The regulation is expected to benefit industrial water users by reducing sediment concentrations in source waters and thus increasing the useful life of industrial equipment.
- Agricultural water use. The regulation is expected to benefit agricultural producers by reducing sediment discharges and, as a result, sediment deposition on farm land; this would lead to improvements in land productivity and enhanced marketability of agricultural products.
- Drinking water treatment.[22] The regulation is expected to reduce the cost of drinking water treatment to improve taste and odor. Reducing nutrient loadings to surface waters is expected to reduce eutrophication which is one of the main causes of taste and odor impairment in drinking water. Taste and odor in drinking water has a major negative impact on the public perception of drinking water safety and the drinking water industry due to a significant increase in drinking water treatment costs from foul taste and odor in the source waters. *Chapter 9* of this report provides detail on eutrophication impacts on portable water.

11.2.4 Limitations Inherent in the Estimate of Nonmarket Benefits

To estimate nonmarket benefits of water quality improvements resulting from the regulation, EPA used a benefits transfer function based on meta-analysis of surface water valuation studies. Key limitations of this approach that are likely to lead to underestimation on nonmarket benefits are outlined below:

- Using benefit transfer from existing surface water valuation studies to estimate nonmarket benefits of the regulation. A number of issues are common to all benefit transfers. Benefit transfer involves adapting research conducted for another purpose in the available literature to address the policy questions at hand. As is typical in applied benefit transfers, the meta-analysis model used in the analysis of C&D benefits provides a close, but not perfect, match to the context in which values are desired. Therefore, some beneficial effects associated with reducing sediment discharges from construction sites are not accounted for in the estimated WTP values. For example, surface water valuation studies included in meta-data focused primarily on changes in ambient water quality. Other effects associated with sediment

[21] The *nonmarket* component (i.e., increased satisfaction with the property) may be implicitly accounted for in WTP for improvements in environmental services provided by surface waters affected by construction site discharges (see *Chapter 10* for detail), however this does not encompass increased *market* values of real estate.

[22] Although EPA estimated a change in drinking water cost associated with reducing turbidity in the source water this estimate does not account for any changes in the cost of treating T/O.

discharges (e.g., sedimentation of river beds, stability and erosion of river banks, and changes in the stream carrying capacity) were not included in the original study scenarios and, as a result, from the estimated WTP for reducing sediment discharges from construction sites.

- Restriction of WTP to the state-level improvements in water quality. In analyzing benefits of the C&D rule, EPA assumed that households would gain no benefits from water quality improvements in aquatic resources located outside of their state of residence. Although empirical literature shows that WTP for environmental quality improvements is inversely related to the distance from the water resource in question, WTP values are likely to be positive for out-of-state residents if improvements occur in aquatic resources of regional or national significance (e.g., Chesapeake Bay, Great Lakes). It is also possible that some of the out-of-state households have non zero WTP for water quality improvements in water resources that have only local significance. Even if per household WTP for out-of-state residents are small they can be very large in the aggregate if these values are held by a substantial fraction of the population.
- Percent of the total reach miles expected to improve maybe understated. Excluding smaller streams from water quality analysis may understate the percentage of both (1) the total reach miles estimated to experience water quality improvements and/or (2) reach miles associated with higher reductions in pollutant concentrations. Because WTP for regional level water quality improvements is a function of the magnitude of the expected water quality change and the geographic scale of improvement (i.e., percent of reach miles expected to improve), the estimated WTP value may be understated. In addition, approximately 67 percent of estuarine/coastal reaches are excluded from the analysis due to data limitations.
- Proximity of water quality improvements to residential areas is not accounted for. Most construction impacts are to surface waters near major population centers. Because people tend to place a higher value on improvements in waterbodies closer to where they live omitting the distance effect from the analysis may lead to understatement of benefits.

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Appendix A – Status of Available Data on Waterbody Assessments

This appendix presents the year of the data available from each state on waterbody assessments. These are the data used in the *National Water Quality Inventory Report to Congress* (USEPA 2009e) and presented in *Sections 2.6* and *3.3.1.2*.

Table A-1: Year of Available Waterbody Assessment Data, as of September 17, 2009

State Name	Assessed Waters Report Year	Impaired Waters Report Year
Alabama	2008	2008
Alaska	2008	2008
American Samoa	2008	2008
Arizona	2008	2004
Arkansas	2004	2004
California	2004	2006
Colorado	2008	2008
Connecticut	2008	2008
Delaware	2006	2006
District Of Columbia	2008	2008
Florida	2002	2002
Georgia	2008	2008
Guam	2008	2008
Hawaii	2006	2006
Idaho	2008	2008
Illinois	2006	2006
Indiana	2008	2008
Iowa	2008	2008
Kansas	2008	2008
Kentucky	2008	2008
Louisiana	2008	2006
Maine	2008	2008
Maryland	2002	2006
Massachusetts	2006	2006
Michigan	2008	2008
Minnesota	2008	2008
Mississippi	2008	2008
Missouri	2008	2008
Montana	2006	2006
N. Mariana Islands	2008	2008
Nebraska	2008	2008
Nevada	2006	2004
New Hampshire	2006	2006
New Jersey	2006	2006
New Mexico	2008	2008
New York	2008	2008
North Carolina	2006	2006
North Dakota	2008	2008
Ohio	2008	2008
Oklahoma	2008	2008
Oregon	2006	2006
Pennsylvania	2006	2004
Puerto Rico	2008	2008
Rhode Island	2008	2008
South Carolina	2008	2008
South Dakota	2008	2008
Tennessee	2008	2008

Texas	2008	2008
Utah	2006	2006
Vermont	2008	2008
Virgin Islands	2008	2008
Virginia	2008	2008
Washington	2008	2008
West Virginia	2006	2006
Wisconsin	2006	2006
Wyoming	2008	2008

Appendix B – List of Federally Threatened and Endangered Aquatic Species Potentially Impacted by Sediment and Turbidity

Table B-1: List of Federal Threatened and Endangered Aquatic Species Potentially Impacted by Sediment

Common name	Scientific Name	Threatened/Endangered Status
Fish		
Ala balik (trout)	*Salmo platycephalus*	Endangered
Alabama cavefish	*Speoplatyrhinus poulsoni*	Endangered
Alabama sturgeon	*Scaphirhynchus suttkusi*	Endangered
Amber darter	*Percina antesella*	Endangered
Apache trout	*Oncorhynchus apache*	Threatened
Arkansas River shiner	*Notropis girardi*	Threatened
Ash Meadows Amargosa pupfish	*Cyprinodon nevadensis mionectes*	Endangered
Ash Meadows speckled dace	*Rhinichthys osculus nevadensis*	Endangered
Asian bonytongue	*Scleropages formosus*	Endangered
Atlantic salmon	*Salmo salar*	Endangered
Ayumodoki (loach)	*Hymenophysa curta*	Endangered
Bayou darter	*Etheostoma rubrum*	Threatened
Beautiful shiner	*Cyprinella formosa*	Threatened
Beluga sturgeon	*Huso huso*	Threatened
Big Bend gambusia	*Gambusia gaigei*	Endangered
Big Spring spinedace	*Lepidomeda mollispinis pratensis*	Threatened
Blackside dace	*Phoxinus cumberlandensis*	Threatened
Blue shiner	*Cyprinella caerulea*	Threatened
Bluemask (=jewel) Darter	*Etheostoma sp.*	Endangered
Bonytail chub	*Gila elegans*	Endangered
Borax Lake chub	*Gila boraxobius*	Endangered
Boulder darter	*Etheostoma wapiti*	Endangered
Bull Trout	*Salvelinus confluentus*	Threatened
Cahaba shiner	*Notropis cahabae*	Endangered
Cape Fear shiner	*Notropis mekistocholas*	Endangered
Catfish	*Pangasius sanitwongsei*	Endangered
Cherokee darter	*Etheostoma scotti*	Threatened
Chihuahua chub	*Gila nigrescens*	Threatened
Chinook salmon	*Oncorhynchus (=Salmo) tshawytscha*	Endangered
Chum salmon	*Oncorhynchus (=Salmo) keta*	Threatened
Cicek (minnow)	*Acanthorutilus handlirschi*	Endangered
Clear Creek gambusia	*Gambusia heterochir*	Endangered
Clover Valley speckled dace	*Rhinichthys osculus oligoporus*	Endangered
Coho salmon	*Oncorhynchus (=Salmo) kisutch*	Endangered
Colorado pikeminnow (=squawfish)	*Ptychocheilus lucius*	Endangered
Comanche Springs pupfish	*Cyprinodon elegans*	Endangered
Conasauga logperch	*Percina jenkinsi*	Endangered
Cui-ui	*Chasmistes cujus*	Endangered
Delta smelt	*Hypomesus transpacificus*	Threatened
Desert dace	*Eremichthys acros*	Threatened
Desert pupfish	*Cyprinodon macularius*	Endangered
Devils Hole pupfish	*Cyprinodon diabolis*	Endangered
Devils River minnow	*Dionda diaboli*	Threatened
Duskytail darter	*Etheostoma percnurum*	Endangered
Etowah darter	*Etheostoma etowahae*	Endangered
Foskett speckled dace	*Rhinichthys osculus ssp.*	Threatened
Fountain darter	*Etheostoma fonticola*	Endangered

Table B-1: List of Federal Threatened and Endangered Aquatic Species Potentially Impacted by Sediment

Common name	Scientific Name	Threatened/Endangered Status
Gila chub	*Gila intermedia*	Endangered
Gila topminnow (incl. Yaqui)	*Poeciliopsis occidentalis*	Endangered
Gila trout	*Oncorhynchus gilae*	Threatened
Goldline darter	*Percina aurolineata*	Threatened
Greenback cutthroat trout	*Oncorhynchus clarki stomias*	Threatened
Gulf sturgeon	*Acipenser oxyrinchus desotoi*	Threatened
Hiko White River springfish	*Crenichthys baileyi grandis*	Endangered
Humpback chub	*Gila cypha*	Endangered
Hutton tui chub	*Gila bicolor ssp.*	Threatened
Ikan temoleh (minnow)	*Probarbus jullieni*	Endangered
Independence Valley speckled dace	*Rhinichthys osculus lethoporus*	Endangered
June sucker	*Chasmistes liorus*	Endangered
Kendall Warm Springs dace	*Rhinichthys osculus thermalis*	Endangered
Lahontan cutthroat trout	*Oncorhynchus clarki henshawi*	Threatened
Leon Springs pupfish	*Cyprinodon bovinus*	Endangered
Leopard darter	*Percina pantherina*	Threatened
Little Colorado spinedace	*Lepidomeda vittata*	Threatened
Little Kern golden trout	*Oncorhynchus aguabonita whitei*	Threatened
Loach minnow	*Tiaroga cobitis*	Threatened
Lost River sucker	*Deltistes luxatus*	Endangered
Maryland darter	*Etheostoma sellare*	Endangered
Mexican blindcat (catfish)	*Prietella phreatophila*	Endangered
Miyako tango (=Toyko bitterling)	*Tanakia tanago*	Endangered
Moapa dace	*Moapa coriacea*	Endangered
Modoc Sucker	*Catostomus microps*	Endangered
Mohave tui chub	*Gila bicolor mohavensis*	Endangered
Nekogigi (catfish)	*Coreobagrus ichikawai*	Endangered
Neosho madtom	*Noturus placidus*	Threatened
Niangua darter	*Etheostoma nianguae*	Threatened
North American green sturgeon	*Acipenser medirostris*	Threatened
Okaloosa darter	*Etheostoma okaloosae*	Endangered
Oregon chub	*Oregonichthys crameri*	Endangered
Owens pupfish	*Cyprinodon radiosus*	Endangered
Owens tui chub	*Gila bicolor snyderi*	Endangered
Ozark cavefish	*Amblyopsis rosae*	Threatened
Pahranagat roundtail chub	*Gila robusta jordani*	Endangered
Pahrump poolfish	*Empetrichthys latos*	Endangered
Paiute cutthroat trout	*Oncorhynchus clarki seleniris*	Threatened
Palezone shiner	*Notropis albizonatus*	Endangered
Pallid sturgeon	*Scaphirhynchus albus*	Endangered
Pecos bluntnose shiner	*Notropis simus pecosensis*	Threatened
Pecos gambusia	*Gambusia nobilis*	Endangered
Pygmy madtom	*Noturus stanauli*	Endangered
Pygmy Sculpin	*Cottus paulus (=pygmaeus)*	Threatened
Railroad Valley springfish	*Crenichthys nevadae*	Threatened
Razorback sucker	*Xyrauchen texanus*	Endangered
Relict darter	*Etheostoma chienense*	Endangered
Rio Grande silvery minnow	*Hybognathus amarus*	Endangered
Roanoke logperch	*Percina rex*	Endangered
San Marcos gambusia	*Gambusia georgei*	Endangered
Santa Ana sucker	*Catostomus santaanae*	Threatened
Scioto madtom	*Noturus trautmani*	Endangered
Shortnose sturgeon	*Acipenser brevirostrum*	Endangered
Shortnose Sucker	*Chasmistes brevirostris*	Endangered
Slackwater darter	*Etheostoma boschungi*	Threatened
Slender chub	*Erimystax cahni*	Threatened

Table B-1: List of Federal Threatened and Endangered Aquatic Species Potentially Impacted by Sediment

Common name	Scientific Name	Threatened/Endangered Status
Smalltooth sawfish	*Pristis pectinata*	Endangered
Smoky madtom	*Noturus baileyi*	Endangered
Snail darter	*Percina tanasi*	Threatened
Sockeye salmon	*Oncorhynchus (=Salmo) nerka*	Endangered
Sonora chub	*Gila ditaenia*	Threatened
Spikedace	*Meda fulgida*	Threatened
Spotfin Chub	*Erimonax monachus*	Threatened
Steelhead	*Oncorhynchus (=Salmo) mykiss*	Endangered
Thailand giant catfish	*Pangasianodon gigas*	Endangered
Tidewater goby	*Eucyclogobius newberryi*	Endangered
Topeka shiner	*Notropis topeka (=tristis)*	Endangered
Totoaba (seatrout or weakfish)	*Cynoscion macdonaldi*	Endangered
Unarmored threespine stickleback	*Gasterosteus aculeatus williamsoni*	Endangered
Vermilion darter	*Etheostoma chermocki*	Endangered
Virgin River Chub	*Gila seminuda (=robusta)*	Endangered
Waccamaw silverside	*Menidia extensa*	Threatened
Warm Springs pupfish	*Cyprinodon nevadensis pectoralis*	Endangered
Warner sucker	*Catostomus warnerensis*	Threatened
Watercress darter	*Etheostoma nuchale*	Endangered
White River spinedace	*Lepidomeda albivallis*	Endangered
White River springfish	*Crenichthys baileyi baileyi*	Endangered
White sturgeon	*Acipenser transmontanus*	Endangered
Woundfin	*Plagopterus argentissimus*	Endangered
Yaqui catfish	*Ictalurus pricei*	Threatened
Yaqui chub	*Gila purpurea*	Endangered
Yellowfin madtom	*Noturus flavipinnis*	Threatened
Amphibians		
African viviparous toads	*Nectophrynoides spp.*	Endangered
Arroyo (=arroyo southwestern) toad	*Bufo californicus (=microscaphus)*	Endangered
Barton Springs salamander	*Eurycea sosorum*	Endangered
California red-legged frog	*Rana aurora draytonii*	Threatened
California tiger Salamander	*Ambystoma californiense*	Endangered
California tiger Salamander (Sonoma)	*Ambystoma californiense*	Endangered
Cameroon toad	*Bufo superciliaris*	Endangered
Cheat Mountain salamander	*Plethodon nettingi*	Threatened
Chinese giant salamander	*Andrias davidianus (=davidianus d.)*	Endangered
Chiricahua leopard frog	*Rana chiricahuensis*	Threatened
Desert slender salamander	*Batrachoseps aridus*	Endangered
Frosted Flatwoods salamander	*Ambystoma cingulatum*	Threatened
Golden coqui	*Eleutherodactylus jasperi*	Threatened
Goliath Frog	*Conraua goliath*	Threatened
Guajon	*Eleutherodactylus cooki*	Threatened
Houston toad	*Bufo houstonensis*	Endangered
Israel painted frog	*Discoglossus nigriventer*	Endangered
Japanese giant salamander	*Andrias japonicus (=davidianus j.)*	Endangered
Mississippi gopher Frog	*Rana capito sevosa*	Endangered
Monte Verde golden toad	*Bufo periglenes*	Endangered
Mountain yellow-legged frog	*Rana muscosa*	Endangered
Panamanian golden frog	*Atelopus varius zeteki*	Endangered
Puerto Rican crested toad	*Peltophryne lemur*	Threatened
Red Hills salamander	*Phaeognathus hubrichti*	Threatened
Reticulated flatwoods salamander	*Ambystoma bishopi*	Endangered
San Marcos salamander	*Eurycea nana*	Threatened
Santa Cruz long-toed salamander	*Ambystoma macrodactylum croceum*	Endangered
Shenandoah salamander	*Plethodon shenandoah*	Endangered
Sonora tiger Salamander	*Ambystoma tigrinum stebbinsi*	Endangered

Table B-1: List of Federal Threatened and Endangered Aquatic Species Potentially Impacted by Sediment

Common name	Scientific Name	Threatened/Endangered Status
Stephen Island frog	*Leiopelma hamiltoni*	Endangered
Texas blind salamander	*Typhlomolge rathbuni*	Endangered
Wyoming Toad	*Bufo baxteri (=hemiophrys)*	Endangered
Panamanian golden frog	*Atelopus varius zeteki*	Endangered
Mollusks		
Cumberland elktoe	*Alasmidonta atropurpurea*	Endangered
Dwarf wedgemussel	*Alasmidonta heterodon*	Endangered
Appalachian elktoe	*Alasmidonta raveneliana*	Endangered
Fat three-ridge (mussel)	*Amblema neislerii*	Endangered
Ouachita rock pocketbook	*Arkansia wheeleri*	Endangered
Birdwing pearlymussel	*Conradilla caelata*	Endangered
Fanshell	*Cyprogenia stegaria*	Endangered
Tampico pearlymussel	*Cyrtonaias tampicoensis tecomatensis*	Endangered
Dromedary pearlymussel	*Dromus dromas*	Endangered
Chipola slabshell	*Elliptio chipolaensis*	Threatened
Tar River spinymussel	*Elliptio steinstansana*	Endangered
Purple bankclimber (mussel)	*Elliptoideus sloatianus*	Threatened
Cumberlandian combshell	*Epioblasma brevidens*	Endangered
Oyster mussel	*Epioblasma capsaeformis*	Endangered
Curtis pearlymussel	*Epioblasma florentina curtisii*	Endangered
Yellow blossom (pearlymussel)	*Epioblasma florentina florentina*	Endangered
Tan riffleshell	*Epioblasma florentina walkeri (=E. walkeri)*	Endangered
Upland combshell	*Epioblasma metastriata*	Endangered
Catspaw (=purple cat's paw pearlymussel)	*Epioblasma obliquata obliquata*	Endangered
White catspaw (pearlymussel)	*Epioblasma obliquata perobliqua*	Endangered
Southern acornshell	*Epioblasma othcaloogensis*	Endangered
Southern combshell	*Epioblasma penita*	Endangered
Green blossom (pearlymussel)	*Epioblasma torulosa gubernaculum*	Endangered
Northern riffleshell	*Epioblasma torulosa rangiana*	Endangered
Tubercled blossom (pearlymussel)	*Epioblasma torulosa torulosa*	Endangered
Turgid blossom (pearlymussel)	*Epioblasma turgidula*	Endangered
Shiny pigtoe	*Fusconaia cor*	Endangered
Finerayed pigtoe	*Fusconaia cuneolus*	Endangered
Cracking pearlymussel	*Hemistena lata*	Endangered
Pink mucket (pearlymussel)	*Lampsilis abrupta*	Endangered
Finelined pocketbook	*Lampsilis altilis*	Threatened
Higgins eye (pearlymussel)	*Lampsilis higginsii*	Endangered
Orangenacre mucket	*Lampsilis perovalis*	Threatened
Arkansas fatmucket	*Lampsilis powellii*	Threatened
Speckled pocketbook	*Lampsilis streckeri*	Endangered
Shinyrayed pocketbook	*Lampsilis subangulata*	Endangered
Alabama lampmussel	*Lampsilis virescens*	Endangered
Carolina heelsplitter	*Lasmigona decorata*	Endangered
Scaleshell mussel	*Leptodea leptodon*	Endangered
Louisiana pearlshell	*Margaritifera hembeli*	Threatened
Alabama moccasinshell	*Medionidus acutissimus*	Threatened
Coosa moccasinshell	*Medionidus parvulus*	Endangered
Gulf moccasinshell	*Medionidus penicillatus*	Endangered
Ochlockonee moccasinshell	*Medionidus simpsonianus*	Endangered
Nicklin's pearlymussel	*Megalonaias nicklineana*	Endangered
Ring pink (mussel)	*Obovaria retusa*	Endangered
Littlewing pearlymussel	*Pegias fabula*	Endangered
White wartyback (pearlymussel)	*Plethobasus cicatricosus*	Endangered
Orangefoot pimpleback (pearlymussel)	*Plethobasus cooperianus*	Endangered
Clubshell	*Pleurobema clava*	Endangered
James spinymussel	*Pleurobema collina*	Endangered

Table B-1: List of Federal Threatened and Endangered Aquatic Species Potentially Impacted by Sediment

Common name	Scientific Name	Threatened/Endangered Status
Black clubshell	*Pleurobema curtum*	Endangered
Southern clubshell	*Pleurobema decisum*	Endangered
Dark pigtoe	*Pleurobema furvum*	Endangered
Southern pigtoe	*Pleurobema georgianum*	Endangered
Cumberland pigtoe	*Pleurobema gibberum*	Endangered
Flat pigtoe	*Pleurobema marshalli*	Endangered
Ovate clubshell	*Pleurobema perovatum*	Endangered
Rough pigtoe	*Pleurobema plenum*	Endangered
Oval pigtoe	*Pleurobema pyriforme*	Endangered
Heavy pigtoe	*Pleurobema taitianum*	Endangered
Fat pocketbook	*Potamilus capax*	Endangered
Alabama (=inflated) heelsplitter	*Potamilus inflatus*	Threatened
Triangular Kidneyshell	*Ptychobranchus greenii*	Endangered
Rough rabbitsfoot	*Quadrula cylindrica strigillata*	Endangered
Winged Mapleleaf	*Quadrula fragosa*	Endangered
Cumberland monkeyface (pearlymussel)	*Quadrula intermedia*	Endangered
Appalachian monkeyface (pearlymussel)	*Quadrula sparsa*	Endangered
Stirrupshell	*Quadrula stapes*	Endangered
Pale lilliput (pearlymussel)	*Toxolasma cylindrellus*	Endangered
Purple bean	*Villosa perpurpurea*	Endangered
Cumberland bean (pearlymussel)	*Villosa trabalis*	Endangered
Crustaceans		
Madison Cave isopod	*Antrolana lira*	Threatened
Conservancy fairy shrimp	*Branchinecta conservatio*	Endangered
Longhorn fairy shrimp	*Branchinecta longiantenna*	Endangered
Vernal pool fairy shrimp	*Branchinecta lynchi*	Threatened
San Diego fairy shrimp	*Branchinecta sandiegonensis*	Endangered
Cave crayfish	*Cambarus aculabrum*	Endangered
Cave crayfish	*Cambarus zophonastes*	Endangered
Illinois cave amphipod	*Gammarus acherondytes*	Endangered
Noel's Amphipod	*Gammarus desperatus*	Endangered
Vernal pool tadpole shrimp	*Lepidurus packardi*	Endangered
Lee County cave isopod	*Lirceus usdagalun*	Endangered
Nashville crayfish	*Orconectes shoupi*	Endangered
Shasta crayfish	*Pacifastacus fortis*	Endangered
Squirrel Chimney Cave shrimp	*Palaemonetes cummingi*	Threatened
Alabama cave shrimp	*Palaemonias alabamae*	Endangered
Kentucky cave shrimp	*Palaemonias ganteri*	Endangered
Kauai cave amphipod	*Spelaeorchestia koloana*	Endangered
Riverside fairy shrimp	*Streptocephalus woottoni*	Endangered
Peck's cave amphipod	*Stygobromus (=Stygonectes) pecki*	Endangered
Hay's Spring amphipod	*Stygobromus hayi*	Endangered
California freshwater shrimp	*Syncaris pacifica*	Endangered
Socorro isopod	*Thermosphaeroma thermophilus*	Endangered
Corals		
Coral, elkhorn	*Acropora palmata*	Threatened
Coral, staghorn	*Acropora cervicornis*	Threatened

Source: USFWS 2009

Appendix C – SPARROW Model Documentation

C.1 General Overview

SPARROW (**SPA**tially **R**eferenced **R**egressions **O**n **W**atershed attributes) is a watershed modeling technique for estimating contaminant source contributions and transport in watersheds and surface waters. SPARROW employs a statistically estimated nonlinear regression model with contaminant supply and process components, including surface-water flow paths, non-conservative transport processes, and mass-balance constraints. Regression equation parameters are estimated by correlating stream water-quality records with GIS (Geographic Information System) data on pollutant sources (atmospheric, point, nonpoint) and climatic and hydrogeologic properties (e.g., precipitation, topography, vegetation, soils, water routing) that affect contaminant transport. The procedure for estimating SPARROW parameters also provides measures of uncertainty in model coefficients and in water-quality predictions. The SPARROW software is written in SAS (Statistical Analysis System) IML (Interactive Matrix Language).

SPARROW model infrastructure consists of a detailed watershed – stream reach network to which all monitoring stations and GIS data on watershed properties are spatially referenced. Digital elevation models (DEMs) are used to delineate watershed boundaries and to identify overland flowpaths. The spatially distributed model structure allows separate statistical estimation of land- and water-related parameters that quantify rates of pollutant delivery from sources to streams, and transport of pollutants to downstream locations within the stream network (i.e., reaches, reservoirs and estuaries). Mechanistic separation of terrestrial and aquatic features of large watersheds, and improved parameter estimation techniques represent significant advances in water-quality modeling to objectively evaluate alternative hypotheses about important contaminant sources and watershed properties that control transport over large spatial scales.

SPARROW has been applied in the analysis of suspended sediment, surface-water nutrients, pesticides, organic carbon, and fecal bacteria, and is potentially applicable to other measures of water quality. Recent applications of SPARROW have provided estimates of nutrient sources and long-term rates of nutrient removal in surface waters (e.g., Smith et al. 1997; Alexander et al. 2000, 2001, 2002a, 2002b). The model has demonstrated particular utility for quantifying long-distance transport and delivery of nutrients to sensitive downstream locations (e.g., estuaries, reservoirs, drinking water intakes). The earliest version of the SPARROW model was developed to describe contaminant transport in surface waters of the State of New Jersey (Smith et al. 1994). Federal and State environmental managers are currently using SPARROW to assess the sources of nutrient loadings in streams, including targeting of nutrient reduction strategies in the Chesapeake Bay watershed (Preston and Brakebill 1999) and in waters of the State of Kansas (Kansas Dept. Health and Environment 2004) as well as for developing TMDLs (Total Maximum Daily Loads) in the Connecticut River Basin (NEIWPCC 2004). Other applications encompass New England watersheds (Moore et al. 2004), New Zealand river basins (Alexander et al. 2002a; Elliott et al. 2005), North Carolina coastal watersheds (McMahon et al. 2003), and watersheds in Tennessee and Kentucky (Hoos 2005). SPARROW models are currently under development for the Delaware River Basin and are being planned for selected regions of the U.S. under the U.S. Geological Survey (USGS) National Water-Quality Assessment (NAWQA) Program.

C.2 Modeling Concept

C.2.1 Objectives of SPARROW Modeling

The broad objective of SPARROW modeling is to establish a mathematical relation between water-quality measurements made at a network of monitoring stations and attributes of the watersheds containing the stations. Once constructed, the model may be used to satisfy a variety of water-quality information objectives. One common modeling objective is to describe past or present water-quality conditions for a state or region on the basis of monitoring data. The underlying challenge is to extrapolate from a sample of water-quality measurements made at a finite number of stream and river locations to an area containing un-sampled locations. The usual limitations in attempting this are: (1) sparse sampling, reflecting the high cost of monitoring; and/or (2) unrepresentative (nonrandom or targeted) sampling undertaken to characterize water quality at specific locations, especially those suspected of having water-quality problems. In the absence of an interpretive model such as SPARROW, a single monitoring design cannot be optimal for both of these distinct objectives. Because the Federal Clean Water Act requires state governments to collect and report both types of information, the distinction between the two types of monitoring is of great practical importance. “Probabilistic” monitoring has been promoted by the U.S. Environmental Protection Agency (EPA) (Yoder 1997) to obtain a spatially unbiased, broad overview of water-quality conditions in State waters. An important limitation of this approach is that the monitoring data alone do not provide detailed information on the geography of water-quality conditions and give little understanding of the factors (sources and processes) that explain those conditions. Targeted monitoring has been used extensively by the States to identify specific streams with water-quality problems and to gage compliance with State water-quality regulatory standards and criteria. These data, however, provide a spatially biased description of water-quality conditions in watersheds. SPARROW is effective in integrating samples from these different monitoring approaches to provide both a geographically representative description of water-quality conditions as well as insight into the sources and watershed processes that control water quality.

A second objective of SPARROW modeling is to identify and quantify the sources of pollution that give rise to in-stream water-quality conditions. SPARROW distinguishes between source categories, such as point sources, atmospheric sources, and agriculture; and individual sources, defined as the rate of supply of contaminant of a particular category originating in the watershed and draining to a specific stream reach. The ability to develop quantitative information on pollution sources in SPARROW models stems from the ability to trace, for each contaminant category, the predicted in-stream flux through a given stream reach to the individual sources in each of the upstream reach watersheds contributing contamination to that reach. Sources may be quantified either in mass units or in terms of their percent contribution to the total contaminant flux to the reach. An example application of SPARROW in quantifying pollution sources is in TMDL analyses (e.g., McMahon and Roessler 2002). In general, the Clean Water Act requires TMDL analyses for any stream reach in which the concentration of a contaminant exceeds the applicable water-quality standard when all pollution discharge limits are met. The ultimate objective of TMDL-related modeling is to establish a hypothetical waste-load allocation for all individual sources affecting the reach in question that would meet the standard. In theory, an infinite number of hypothetical load allocations will satisfy the standard, and choosing the official allocation requires comparing many possible solutions, utilizing simulation (see below) in search of a least-cost or other optimum solution. Prior to conducting the hypothetical analyses, however, a great deal of preliminary quantitative information on the actual or baseline relations between watershed sources and in-stream conditions is useful. The percentage contributions (shares) of individual point sources are of particular interest because point sources are usually the only sources subject to direct regulation. The share contributions of individual nonpoint sources are also of interest as a means of identifying the important stakeholders to include in discussions of voluntary pollution reductions. This describes the current application of SPARROW: to evaluate the effectiveness of a range of regulatory options for controlling and mitigating sediments associated with construction activities, with explicit focus on the impacts of such options on in-stream water quality as measured by Total Suspended Solids (TSS).

A third objective is water quality simulation. Simulation refers to the use of a calibrated model to predict conditions on the basis of a set of altered, typically hypothetical model inputs. The ability to portray counterfactual conditions for specified inputs is one of the most powerful justifications for models; there are often no alternative methods for conducting controlled experiments on complex systems. Water-quality model simulations can depict the in-stream effects of changes in contaminant sources associated with alternative future pollution-control strategies, providing a critical step in the analysis of the costs and benefits associated with specific Best Management Practices (BMPs) or regulatory approaches. Simulating alternative waste-load allocations in a TMDL analysis (see above) is a prime example of a water-quality simulation application. In the present application, SPARROW simulation is used to conduct experiments focused on specific regulatory options (BMPs) for managing construction site sediment that would not be feasible to conduct physically outside a limited number of test sites.

A fourth objective of SPARROW modeling is hypothesis testing. One common feature of the preceding modeling objectives is that they each make use of predictions of the dependent variable by a calibrated model. Another class of modeling objectives focuses on the calibration process and its results directly. The SPARROW estimation process explores the predictive value of a set of potential explanatory variables and may also compare alternative mathematical forms. The selection of a final set of predictors and mathematical form usually has the primary objective of maximizing the accuracy of model predictions of the dependent variable, but an alternative modeling objective may be to test one or more hypotheses about the nature and importance of factors and processes that may have influenced water quality at the locations where samples were collected. Hypothesis tests are performed for each of the model parameters estimated in the calibration, and these serve as indicators of an empirical relation between the independent variables associated with each parameter and the dependent variable of the model. Because the coefficients in a SPARROW model are specified to conform to physical processes, and the potential explanatory variables are selected on the basis of theoretical or logical connections to the dependent variable, a statistically strong parameter provides evidence of the physical relationship. For example, if multiple categories of potential sources of the contaminant are being evaluated, the model estimation process provides useful hypothesis tests on the importance of each category. “Importance” is measured by the correlation between contaminant inputs from the source category and downstream monitored loads of the contaminant. That correlation will tend to be stronger when the mass contribution of the category to the total mass of contaminant flowing past many of the monitoring stations is large, but model estimation may also indicate a source category is important even when the mass contribution is small, provided the amount contributed to stream loads by each unit of source is consistent from place to place.

The importance of factors and processes potentially related to the transport of contaminants from sources to stream channels and within stream channels may be tested through calibration of the “land to water” and “in-stream decay” terms in the model. These terms dictate the fraction of the contaminant mass that completes the terrestrial and aquatic phases of transport within the watershed draining to each stream reach. The land-to-water terms describe the land-surface characteristics that influence both overland and subsurface transport from sources to stream channels. Similarly, the in-stream decay terms describe the effects of channel characteristics on downstream transport. In sum, an important objective in building and calibrating SPARROW models is to gain insight and to test hypotheses concerning the role of specific contaminant sources and hydrologic processes in supplying and transporting contaminants in watersheds.

C.2.2 SPARROW Mass Balance Approach

SPARROW models constructed to date, like most water-quality models, are expressed in the form of a mass balance. Such models describe the movement of mass in space and/or the change of mass in time. The law of conservation of mass implies that certain basic accounting rules must apply to a mass balance, water-quality model, such as: (1) the sum of fluxes entering the confluence of two streams equals the flux leaving the confluence; (2) the sum of the fluxes attributable to each contaminant source must equal total flux; and (3) a doubling of all sources in the model results in an exact doubling of the predicted flux at each location. Because the dependent variable in SPARROW models (the mass of contaminant that passes a specific stream location per unit time) is, in mathematical terms, linearly related to all sources of contaminant mass in the model, all accounting rules relating to the conservation of mass will apply. Mass accounting in SPARROW models is also supported by the explicit spatial structure defined by the stream network.

There are a number of advantages to the mass balance approach. Because of the linear relation between flux and sources, there is an expectation that the estimation of flux over spatial scales smaller and larger than that covered by the model's sample data will yield reasonably accurate results. The imposition of mass balance greatly improves the interpretability of model coefficients. For example, assuming mass balance, the coefficient associated with the reach time-of-travel variable is interpreted as a first-order decay rate and, because point-source loadings are delivered directly to the stream network, a reasonable null hypothesis for coefficients associated with point sources is that they equal 1.0 (Smith et al. 1997). The enhanced interpretability of the model coefficients in turn facilitates the comparison of coefficient estimates from the model with other estimates described in the literature. These comparisons have been generally favorable, especially for the model components that quantify in-stream nitrogen decay rates (Alexander et al. 2000, 2002a; McMahon et al. 2003), nutrient and suspended sediment removal rates in reservoirs (Schwarz et al. 2001; Alexander et al. 2002a), and the nutrient export associated with various land uses and pollutant sources (e.g., Alexander et al. 2001, 2002a, 2004; McMahon et al. 2003). Mass balance provides a basis for flux accounting, whereby flux can be allocated to its various sources, both spatially and topically. For example, mass balance makes it possible to attribute nutrients discharged to the Gulf of Mexico to specific sources within the Mississippi basin (Alexander et al. 2000), thereby providing guidance in managing the reduction of this discharge. This approach is also sensitive to the effects of natural and human-related processes that supply and remove contaminants from watersheds over long periods. Thus, this approach de-emphasizes the quantification of short-term cycling and transformation processes, which are often central to the functioning of many dynamic mechanistic models, in favor of processes that have a long-term impact on elemental budgets in aquatic ecosystems.

C.2.3 Time and Space Scales of the Model

SPARROW models are structurally designed to explain spatial variability in the long-term mean-annual or mean-seasonal flux of contaminants in streams. Spatial variability is modeled as a function of natural and human-related properties of watersheds that influence the supply and transport of contaminants. Estimates of the long-term mean flux are developed from water-quality and streamflow monitoring data that are regularly collected at fixed locations on streams and rivers. The basic form of SPARROW models is structured to describe the long-term, steady-state water-quality and flow conditions in streams.

Contaminant source inputs are assumed to be in balance with the estimated sinks and measured in-stream water-quality load such that there is conservation of mass among the model components. The principal objective supported by this model structure is the quantification of the location and rates (and statistical uncertainties) of the supply, transport, and fate of contaminants within the terrestrial and aquatic ecosystems of watersheds. In the current specifications of SPARROW models, temporal variability in contaminant loads, including intra- and inter-annual variations in water quality and streamflow reflected in the monitoring data, are explicitly modeled and accounted for in a step prior to modeling spatial variability in loads with SPARROW.

The computation of mean-annual or mean-seasonal flux (the SPARROW response variable) requires the prior application of a water-quality flux-estimation model constructed on the basis of streamflow and water-quality records from regularly monitored stream locations. The flux-estimation model explicitly accounts for temporal variability in contaminant loads related to streamflow, season of the year, and trends (either continuous or abrupt) with time. A base-year load estimate ensures that the stream water-quality loads and the contaminant source data (which are commonly reported only periodically, e.g., the U.S. Agricultural Census reports every five years) are contemporaneous. Therefore, the mean-annual loads used to calibrate SPARROW models describe the mean load that would be expected to occur during a particular base year under long-term mean streamflow conditions.

The steady-state mass-balance structure of the basic SPARROW model quantifies the long-term net effects of biogeochemical and hydrologic processes on contaminant transport in terrestrial and aquatic ecosystems. Modeling the effects of these processes is typically of greatest interest for non-conservative chemical, physical, and biological properties of water, such as nutrients, pesticides, fecal coliform bacteria, organic carbon, and suspended sediment. Many of these constituents are subject to chemical transformations or degradation during transport and may be stored over short or long periods. In SPARROW models that are estimated using long-term water-quality records, biogeochemical cycling and storage processes that temporarily immobilize or remove contaminants from flow paths are generally in steady state with those processes that mobilize or release contaminants from storage. Hence, the effects of transformation and removal processes that operate on relatively short intra-annual time scales (e.g., daily, weekly, seasonal), or over multi-year time scales that are less frequent than the period of model estimation, are not likely to be detected as contaminant losses in the steady-state form of SPARROW models. Limitations in our knowledge of temporal lags in chemical transport and their causes also create uncertainties in the periods over which steady-state conditions apply. For example, the time scales for the transport of sediment in streams reflect erosion, storage, and transport processes that operate from seasonal to decadal or even longer (e.g., century) periods (Trimble and Crosson 2000).

Temporal changes in mean conditions also can be explicitly modeled in SPARROW using the current model structure and software. One approach is to estimate multiple steady-state models that explain mean-annual or mean-seasonal stream contaminant loads for separate multi-year periods. Alternatively, the current software permits the estimation of a single SPARROW model having time-dimensioned dependent and independent variables – each assumed to pertain to different steady-state conditions. In this approach, temporal changes in mean-annual stream contaminant flux may be modeled as a function of temporal changes in contaminant sources, land use, and climatic factors (e.g., precipitation, streamflow, and temperature) over similar multi-year periods as those used to estimate stream loads at each monitoring station. The coefficients in this model can be time dependent or restricted to take common values over the full period of the analysis. This type of model structure has considerable data demands that require the development of historical data on contaminant sources and climatic/hydrologic variability in the watersheds and reaches (including stream velocity, which is sensitive to average streamflow and thus varies in response to changes in mean streamflow across different periods). Such a model allows users to explicitly test for temporal changes in model parameters to determine whether changes are evident in process rates or contaminant export coefficients with time. As an alternative to the explicit estimation of a SPARROW model with time-dimensioned coefficients, existing SPARROW models can be used to simulate changes in mean-annual stream water-quality loads as a function of changes in source inputs. This approach assumes that the process rates and contaminant export coefficients of the model do not change with time.

C.2.4 Accuracy and Complexity of SPARROW Models

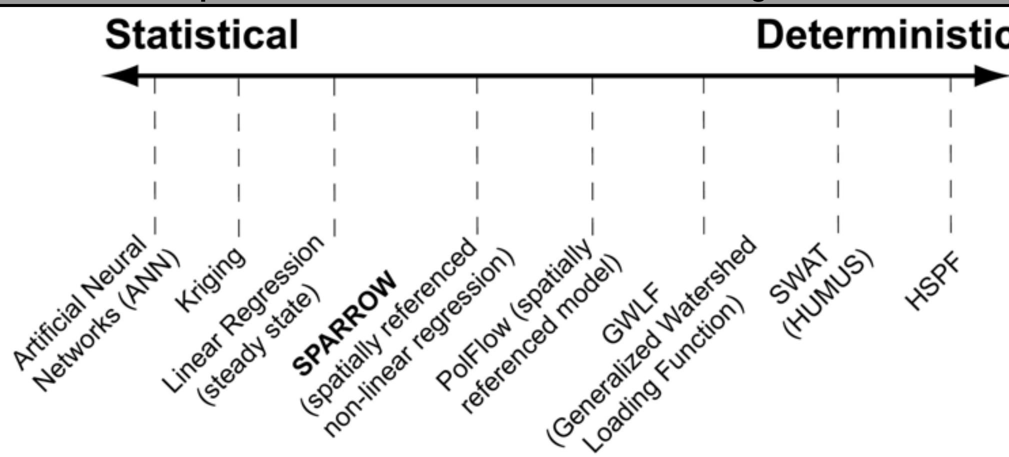
It is generally the case for SPARROW, as for any model with statistically estimated parameters, that model accuracy (bias and precision) and complexity (number of statistically significant or sensitive parameters) are dependent on the information content of the water-resources data used in model calibration. Investigations of hydrologic models have demonstrated that both the *quantity* and *quality* of calibration data define the information content and have important effects on parameter estimation and precision (e.g., Gupta and Sorooshian 1985; Yapo et al. 1996). Increasing the *quantity* of data can improve the precision provided the data give new, independent information about the values of the model parameters. Data *quality*, as defined by Gupta and Sorooshian (1985), generally increases as the data become more "representative" of the range of watershed properties that affect transport and the range of conditions present in the sampled watersheds. An independent set of measurements are preferred for estimating parameter values, such that the data reflect the most extreme combinations of watershed conditions for the various properties. These general statistical guidelines have implications for the time and space scales required to develop SPARROW data sets and accurate models. First, a sufficiently large number of water-quality monitoring stations are required. In SPARROW models, the monitoring-station loads serve as the response variable observations in the nonlinear spatial regressions. The number of stations has a demonstrable effect on the statistical power of the regression – i.e., the capacity of the model to detect the effect of an explanatory factor on stream loads. Models with more stations generally have greater power, which typically supports more complex models – i.e., models with a larger number of statistically significant parameters and functional components. Second, the amount of spatial variability in the stream monitoring data and explanatory factors should reflect as broad and representative a range of watershed conditions as possible. The most complex SPARROW models typically have been developed for regions that have relatively large spatial variability (greater than one order of magnitude) in the watershed properties that affect contaminant transport. Watershed properties that vary over a wide range within a modeled region generally provide more information about the response of stream loads to different levels of a given watershed property and are more likely to be statistically significant in SPARROW models.

The spatial variability of a given variable is most readily increased in SPARROW models by expanding the spatial domain of the model to include larger drainage areas. This, of course, has the effect of increasing the number of monitoring stations, which also contributes to the potential for greater model complexity. In selecting water-quality monitoring sites for modeling, it is especially important to obtain sites that are located on a wide range of stream sizes (especially small streams) and are inclusive of impoundments of varying size. Sediment and nutrient removal rates are highly responsive to the hydraulic characteristics of streams and reservoirs, such as flow volume and velocity (Alexander et al. 2000, 2002a). Accounting for the wide variation in these properties can provide more accurate estimates of nutrient removal and improve the separation of land and water effects on transport in the model. Estimation errors are much larger for constituents that are most affected by high flows, such as suspended sediment, total phosphorus, and fecal coliform bacteria. These constituents are generally more difficult to measure and exhibit larger variability in concentrations in streams, which can produce less precise estimates of the mean-annual flux. By contrast, dissolved substances, such as nitrate, sulfate, and total dissolved solids exhibit less variability with flow, and their fluxes can generally be more accurately estimated.

C.2.5 Comparison of SPARROW with Other Watershed Models

A wide variety of hydrologic and water-quality models have been used to describe contaminant sources and transport in watersheds and surface waters. These models can be characterized on the basis of their process complexity and temporal and spatial scales (e.g., Singh 1995). The level of complexity or process detail represented by model descriptions of hydrologic and biogeochemical processes commonly varies with the extent to which *deterministic* (mechanistic) and *statistical/empirical* methods are used to describe and estimate these processes *(Figure 12-1;* Alexander et al. 2002b). All models reflect some blend of these methods, but most place greater emphasis on one or the other type of model structure and process specification.

Figure 12-1: A Simple Continuum of Model Types Based on the Level of Statistical and Mechanistic Descriptions of Contaminant Sources and Biogeochemical Processes



In general, purely statistical models tend to reflect more simplistic model constructs. These models have a simple correlative mathematical structure and typically assume limited *a priori* knowledge of various processes. Conventional versions of these models are expressed as simple linear (or log linear) correlations of stream measurements with watershed sources and landscape properties (e.g., Peierls et al. 1991; Howarth et al. 1996; Caraco et al. 2003). The methods have the advantage of being readily applied in large watersheds (often relying on generally available stream monitoring records) and can readily quantify the errors in model parameters and predictions. Simple correlative approaches, however, offer little mechanistic explanation of contaminant sources and transport. They generally lack spatial detail on the distribution of sources and sinks within watersheds, rarely account for nonlinear interactions between sources and loss processes, and do not impose mass-balance constraints on contaminant transport. The most purely statistical approaches are found in artificial neural network and kriging techniques. These models commonly provide an excellent fit to the observations, but provide little understanding of the processes that affect contaminant transport.

By contrast, mechanistic water-quality models have complex mass-balance structures that simulate hydrologic and contaminant transport processes, often according to relatively fine temporal scales [e.g., Soil Water Assessment Tool (SWAT), Srinivasan et al. 1993; Agricultural Nonpoint Source Model (AGNPS), Young et al. 1995; Hydrologic Simulation Program-Fortran (HSPF), Bicknell et al. 2001]. The components of these models frequently provide a highly detailed temporal description (e.g., daily, hourly) of the response of stream contaminants to climatic variability, and the effects of coarser temporal variations in land use and management activities are often superimposed on the more detailed climatic variations. The mathematical descriptions of these responses are frequently based on *a priori* assumptions about the dominant processes and their reaction rates. The complexity of mechanistic simulation models creates intensive data and calibration requirements, which generally limits their application to relatively small watersheds. Because mechanistic water-quality models are frequently calibrated manually, robust measures of uncertainty in model parameters and predictions cannot be quantified. Despite the common use of mechanistic models, there are growing concerns about whether sufficient water-resources data and knowledge of biogeochemical processes exist to reliably support the general use of such highly complex descriptions of processes (Jakeman and Hornberger 1993; Beven 2002). Without sufficient data, there is limited ability to apply formal parameter-estimation techniques required to quantify model uncertainties and to identify unique models having sensitive and uncorrelated parameters.

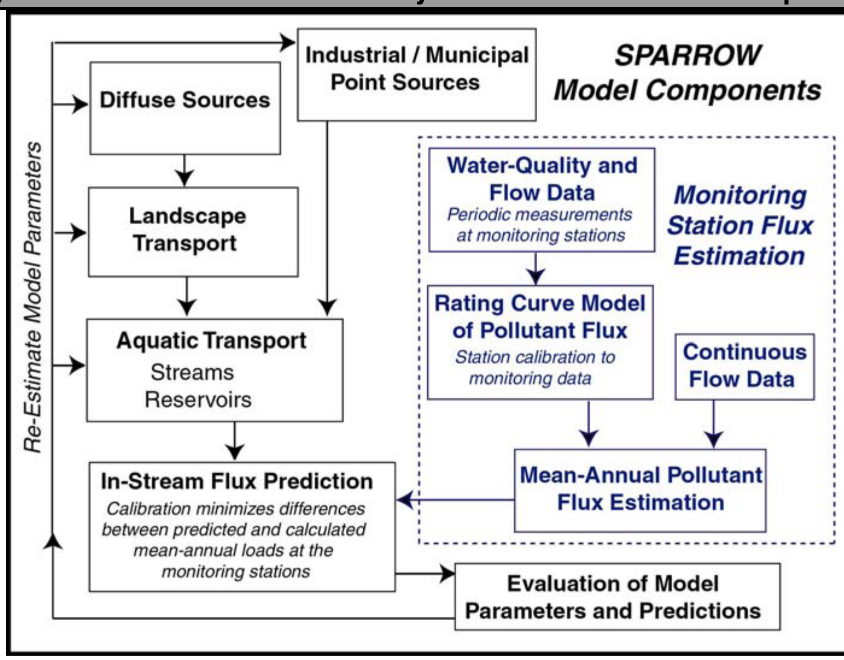
By comparison to other types of water-quality models, SPARROW may be best characterized as a hybrid process-based and statistical modeling approach for estimating pollutant sources and contaminant transport in surface waters. The mechanistic mass transport components of SPARROW include surface-water flow paths (channel time of travel, reservoirs), non-conservative transport processes (i.e., first-order in-stream and reservoir decay), and mass-balance constraints on model inputs (sources), losses (terrestrial and aquatic losses/storage), and outputs (riverine nutrient export). The statistical features of the model involve the use of nonlinear parameter-estimation for spatially correlating stream water-quality records with geographic data on pollutant sources (e.g., atmospheric, point-, nonpoint) and climatic and hydrogeologic properties (e.g., precipitation, topography, vegetation, soils, water routing). Parameter estimation ensures that the calibrated model will not be more complex than can be supported by the data. This provides an objective statistical approach for evaluating alternative hypotheses about important contaminant sources and controlling transport processes over large spatial scales in watersheds. SPARROW has been shown to improve the accuracy and interpretability of model parameters and the predictions of nutrient loadings and sources in streams as compared with conventional statistical modeling approaches (Smith et al. 1997; Alexander et al. 2000, 2002a, 2002b).

A number of inter-model comparisons have been previously conducted that compare the performance and properties of SPARROW to those of other water-quality models. These include national comparisons with SWAT and comparisons in the Chesapeake Bay watershed with HSPF (see Alexander et al. 2001). Inter-model comparisons also have been conducted with statistical and quasi-deterministic models in watersheds of the northeastern U.S. (see Alexander et al. 2002b; Seitzinger et al. 2002). Finally, evaluations of the SPARROW technique also are included as part of a number of recent National Research Council (NRC) reports (2000, 2001a, 2001b, 2002a, 2002b). At least two of the NRC reports (2000, 2001a) have noted the advantages of statistical modeling approaches, such as SPARROW, for general water-quality assessment and use in TMDL assessments.

C.2.6 Model Infrastructure

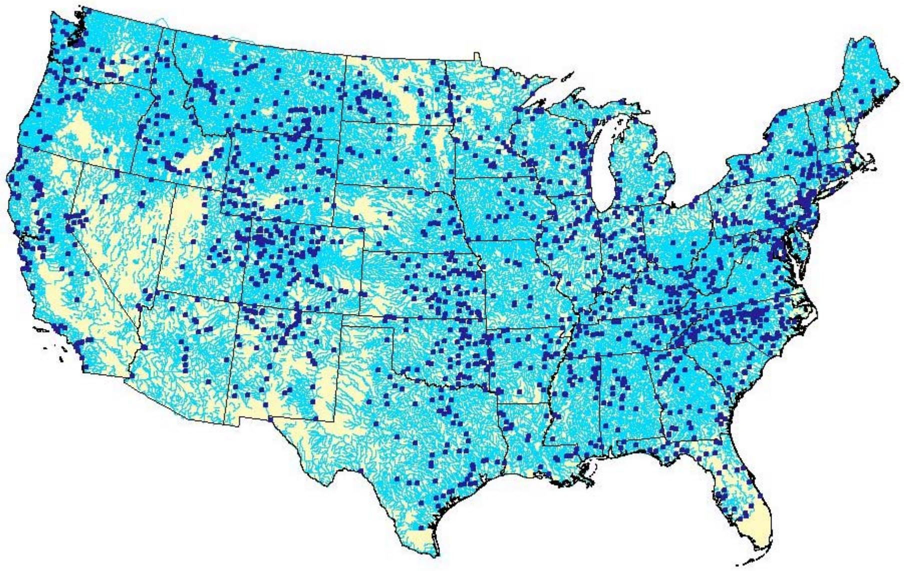
The parameters of SPARROW models are statistically estimated with nonlinear regression techniques by spatially correlating water-quality flux estimates at monitoring stations with watershed data on sources, and landscape and surface-water properties that affect transport. The calibrated models are then used to predict flux, total and disaggregated by contributing source, for stream reaches throughout a river network. A flow diagram is provided in *Figure 12-2* to illustrate the functional linkages between the major spatial components of SPARROW models. Pre-processing steps are required to develop *reach-level* information for the major components of the SPARROW model infrastructure as shown in *Figure 12-3. Monitoring station flux estimation* refers to the estimates of long-term flux used as the response variable in the model. Flux estimates at monitoring stations are derived from station-specific models that relate contaminant concentrations from individual water-quality samples to continuous records of streamflow and time. The *stream reach*, inclusive of its incremental contributing drainage basin, is the most elemental spatial unit of the SPARROW model infrastructure. Stream reaches typically define the length of stream channel that extends from one stream tributary junction to another. Explanatory data (e.g., climate, topography, land use) are frequently compiled according to geographic units that are not coincident with the drainage basin boundaries of river reaches. These data may be collected at different spatial scales and according to spatial units that reflect political (e.g., counties) or other non-hydrologic features of the landscape.

Figure 12-2: Schematic of the Major SPARROW Model Components



Source: Modified from Alexander et al. (2002).

Figure 12-3: Location of 1,828 Water Quality Monitoring Stations Used in the SPARROW Sediment Model, in Relation to the Reach File 1 (RF1) Reach Network



C.2.7 Use of Monitoring Data

The estimation of a SPARROW model requires estimates of long-term mean flux from a spatially distributed set of monitoring stations within the study area, having sufficiently long periods of record. In addition, it is necessary to include data from a diverse set of monitoring stations, inclusive of a wide range of spatial scales and expressing considerable variation in predictor variable conditions. Often, monitoring stations have different periods of record. Long-term variations in hydrologic conditions, combined with long-term trends in water quality, imply mean fluxes computed over different periods may not be directly comparable. If there are trends in water quality, the estimate of long-term mean water-quality flux will depend on the period or window through which the water-quality data are acquired. Additionally, water quality is rarely measured with the frequency necessary to directly estimate long-term mean flux, so that indirect methods are needed to estimate flux from available measurements to account for hydrologic conditions during periods when water-quality measurements are not made. The estimation procedure must account for the fact that water-quality measurements are not always collected in a random manner, implying the arithmetic average of flux is not a reliable estimate of mean flux. Rather, the measurements of flux must be related in some way to other information, collected on a continuous basis, which exhibits a close relation to flux, typically streamflow. The limitation, therefore, in constructing a mass-balance model using long-term mean flux is that only water-quality measurements that can be associated with a continuous record of streamflow will be suitable for model estimation.

Trends in water quality also create problems for the mass balance approach in relating water quality to its predictor variables, principally the source variables. Ideally, the source variables will consist of a time series of estimates, and these source variables would be included in SPARROW as long-term averages in the same way as flux estimates. However, source information is rarely available for multiple periods. Even if source information is available over time, the period over which it is compiled will rarely match the period of the flux information; for example, point-source and land-use data are infrequently available due to the high cost of compiling this information. To address the problem of incompatible periods of record, a SPARROW model is typically specified for a single base year; with all water-quality and source information assumed to pertain to a given point in time. The severest constraint on the period of the analysis is imposed by source information that is available only for a single year. Consequently, the base year is generally selected to be in the middle of the range of available years for this 'one-time' information.

C.2.8 Model Specification for Monitoring Station Flux Estimation

The extrapolation of infrequently sampled water-quality data and removal of trend require the specification of a model of flux. The model must relate infrequently measured data to variables that are measured continuously over time; and to accommodate de-trending, the model must include a function of time. These requirements suggest a model that relates infrequently measured concentration to the variables streamflow and time. The inclusion of streamflow as an explanatory factor serves another purpose in the analysis. Because flux is typically positively related to streamflow, water-quality sampling is commonly biased towards high flow events. The inclusion of flow in the water-quality model effectively conditions the estimation of flux so as to remove the effects of high-flow sampling bias. The estimation of mean flux by a station-specific model need not account for all processes affecting flux within a basin; as this task is assigned to the SPARROW model. All that is required is that the estimated mean flux at a station be reflective of long-term average processes within the basin. Station-specific flux models need not be structurally accurate; they need only be predictively accurate. A causal explanation of flux is ultimately obtained through application of a SPARROW model, whereby variations in mean flux conditions across stations, estimated either explicitly or implicitly, are correlated with variations in basin attributes across space.

Cohn et al. (1992a) have suggested a simple seven-parameter model in which the logarithm of contaminant concentration (c) is related via a linear model to an intercept, the logarithm of flow, the square of the logarithm of flow, decimal time *T*, decimal time squared, and a seasonal harmonic consisting of two trigonometric terms – the sine and cosine of 2 times decimal time. Vecchia (2000) has argued that there are important long-term lags affecting the relation between water quality and flow. He suggests a specification that relates the log of contaminant concentration to a set of compound flow terms consisting of moving averages of flows, of various lengths, in addition to time trend terms. A specification that generalizes both the seven-parameter and the lag flow models takes the form

$$\widetilde{c}_i = M(Q_t)\beta_Q + h(T_t)\beta_T + X_t\beta_X + e_t \qquad \text{(Eq. C-1)}$$

Where Q_t is a *p*-element row vector consisting of current and lagged logarithms of flow, $M(Q)$ is a vector function that transforms the *p*-element logged flows into a K_q element row vector; β_Q is a K_q element vector consisting of coefficients associated with the transformed flow terms; $h(T_t)$ is a K_T element row vector function of decimal time; β_T is a K_T element vector of coefficients associated with the transformed decimal time terms; X_t is a K_X element row vector of other exogenous variables affecting water quality; β_X is a K_X element vector of coefficients associated with the other exogenous variables; and e_t is the normally distributed error term, independent over time and uncorrelated with each of the predictor variables.

C.2.9 Tools for Flux Estimation

The water-quality models described above can be estimated using ordinary least squares if the water-quality data do not include any censored observations. For cases in which observations are censored, Cohn et al. (1992b) suggest estimation via adjusted maximum likelihood. The standard maximum likelihood method for type I censored data (data for which the censoring threshold is known) is the Tobit model. The method of adjusted maximum likelihood combines the Tobit model with an adjustment to correct for first-order bias in the coefficient estimates caused by estimation using a small sample. The method of adjusted maximum likelihood is implemented within the USGS program Load Estimator (LOADEST) 2000 (Runkle et al. 2004). In addition to estimates of the parameters and their covariance matrix, the program uses retransformation methods to produce unbiased estimates of daily and annual flux. A simple averaging of the daily or annual estimates over all days or years yields an estimate of long-term mean flux. The estimation of the model used to detrend flow requires a maximum likelihood method capable of correcting for serial correlation in the errors. This capability is included in the PARMA model developed by Vecchia (2000), and in standard statistical packages such as SAS (SAS 1993). The more recently developed program Fluxmaster (Schwarz et al. 2006) includes methods to estimate the time-series flow model using maximum likelihood, detrend flow, and estimate the water-quality model via adjusted maximum likelihood. The program also computes unbiased, detrended estimates of long-term mean flux, and provides an estimate of the associated standard error. The exact methods used in Fluxmaster differ from those used in LOADEST 2000, but they are a close approximation, and identical if there are no censored observations. Fluxmaster (Schwarz et al. 2006) is used in the present study.

C.2.10 Guidance for Specifying Monitoring Station Flux Models

The ideal water-quality record has sufficient observations to reliably estimate the coefficients in the model. Models that include both flow and time require a fairly long record, one that includes the base date for detrending, and encompassing a wide range of hydrologic conditions. One consideration that places limits on the length of the record is the need for flux estimates to be representative of base-year conditions. This does not imply that the record must be representative of the hydrologic conditions in the base year: the SPARROW model is estimated using mean estimates of water quality with the intention of removing variations due to hydrologic conditions. Rather, long-term patterns in water quality should be adequately captured by the model specification. This implies the period of record should not be so long as to violate the assumption that water-quality trend is reasonably approximated by a simple linear function of time. It is generally true that longer water-quality records display more complex patterns of trend. A record that is too long runs the risk of misrepresenting the trend in water quality for any given year, adding noise to the detrended flux estimate. For national analyses, Smith et al. (1997) and Alexander et al. (2000) have based mean flux on water-quality data that span a 15 to 20 year period. Regional analyses by Preston and Brakebill (1999) and Moore et al. (2004) have used water-quality data that span 20 to 25 years. Shorter periods of about six years were used by Alexander et al. (2002a). Shorter records may certainly be used to obtain an estimate of mean flux, although a shorter record will generally result in a larger standard error in the mean flux estimate. For very short records, however, the specification of trend becomes unreliable. It is recommended that trend terms be excluded from the model if record spans less than three years.

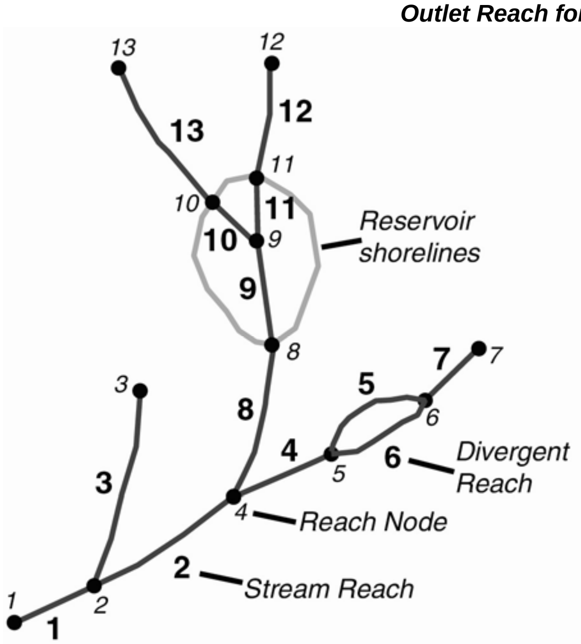
One of the principal determinants of accuracy in flux estimation is the number of observations input to the water-quality model. Generally, it is recommended that an estimate of flux depend on no fewer than 15 uncensored water-quality observations. However, it may be necessary to deviate from this threshold in situations in which a site is subject to high variability in streamflow. The final consideration for inclusion of data for a station in the SPARROW analysis is that the standard error of the mean flux estimate not be too large. SPARROW has the capability of weighting observations according to their standard errors; therefore wholesale exclusion of high-error flux estimates is unnecessary. If, however, a high-error flux estimate enters the model as an upstream source for some downstream flux observation (the nested basin arrangement) the potential for biasing estimates increases. High-error flux estimates also present problems in assessing the error of SPARROW predictions if the predictions are conditioned on observed, upstream flux estimates. In national analyses, we have frequently excluded mean flux estimates that have a standard error greater than about 20 to 30 percent.

C.2.11 Stream Network Topology

A vector- or raster-based digital representation of the stream and river network topology is the most fundamental component of the spatial infrastructure that supports the SPARROW model *(Figure 12-4)*. Vector representations are based on point and line (arc) GIS features, whereas raster representations are based on a cellular (areal) structure. In either instance, a stream reach network explicitly defines surface-water flow paths that spatially connect contaminant sources and landscape features with water quality observations at downstream monitoring stations. In a vector-reach topology, a stream reach represents the length of stream channel that extends from one tributary junction to another. Reach nodes are point features that are associated with the location of tributary junctions, hence reach boundaries. Reach nodes will also occur at the locations where reaches overlay with the shorelines of impoundments (reservoirs, lakes), and with stream water-quality monitoring sites. In a raster representation of streams, nodes may also define various intermediate locations along the stream reach between tributary junctions. Whether digitally represented in vector or raster form, the reach topology must define either a set of reach nodes or raster cells that are linked hydrologically to indicate the direction of water flow. The node topology must be defined according to an upstream (from-) and downstream (to-) node attribute table *(Figure 12-4)*. This tabulation of surface-water flow paths is required for the routing of water and contaminants through the river network by SPARROW navigation software during estimation and application of the model. Thus, the reach-node table defines the fundamental data infrastructure of the model.

Figure 12-4: Schematic Illustrating a Vector Stream Reach Network with Node Topology and Water/Contaminant Reach-Node Routing Table

The reach-type indicator has possible values of "0" (Stream Reach), "1"(Impoundment Reach), and "2" Outlet Reach for Impoundment



Reach-Node Attribute Table

Reach Number	Up-stream Node	Down-stream Node	Diversion Fraction	Reach-Type Indicator
13	13	10	1.0	0
10	10	9	1.0	1
12	12	11	1.0	0
11	11	9	1.0	1
9	9	8	1.0	2
8	8	4	1.0	0
7	7	6	1.0	0
5	6	5	0.7	0
6	6	5	0.3	0
4	5	4	1.0	0
2	4	2	1.0	0
3	3	2	1.0	0
1	2	1	1.0	0

The SPARROW model structure supports distributary and diversion reaches in the stream network; including braided channels or reaches where water is diverted to canals or other water bodies. The

SPARROW model assumes that contaminants are diverted in proportion to volume of flow. Therefore, an estimate is required for each reach of the *diversion fraction* – a measure of the fraction of streamflow that is diverted in distributary reaches. A diversion fraction of 1.0 is assigned to reaches without water diversions. In the example distributary reaches shown in *Figure 12-4*, the fraction of the flow in reach 7 that is diverted to the two downstream reaches is 0.7 for reach 5 and 0.3 for reach 6. The SPARROW reach topology also supports the separate designation of impoundments (e.g., lakes, reservoirs) associated with stream reaches. This designation is used in SPARROW to separately estimate the sediment and contaminant attenuation in reservoirs and lakes. In *Figure 12-4,* reaches 9-11 are associated with a reservoir. A "reach-type" indicator is used to identify reaches associated with impoundments separately from conventional stream reaches. The outlet reach of an impoundment is coded separately from other interior impoundment reaches to facilitate the pollutant attenuation calculations.

In addition to the reach properties listed in *Figure 12-4,* the reach length and an estimate of the mean-annual streamflow of the reach is also required. Estimates of mean water velocity are required to estimate in-stream contaminant attenuation as a function of the water time of travel (alternatively, in-stream attenuation can be estimated as a function the reach length). Measures of the areal water load are needed for impoundments for use in estimating the contaminant attenuation in these water bodies; the areal water load is computed as the quotient of the outflow to surface area of the waterbody (see Alexander et al. 2002a). Digital representations of the drainage basin boundaries associated with river reaches are also needed to estimate the incremental and total drainage area of the reaches and to support the digital overlay of drainage boundaries with polygonal boundaries (vector or raster) that define the locations of contaminant sources (point and diffuse) and various landscape properties.

The assessment of pollutant loadings to coastal estuaries requires an expanded reach network that includes shoreline features and the identification of reaches that terminate at estuaries. The node points of shoreline reaches include the downstream nodes of the terminating reaches. Shoreline reaches are used to define coastal drainage areas – areas that discharge runoff directly to the estuary without transport through a stream reach. Because the discharge for a shoreline reach does not accumulate from any upstream location, the diversion fraction for these features is set to zero. Shoreline reaches also have no stream attenuation so travel time is set to zero.

S*patial referencing* is a critical step in SPARROW modeling that is necessary to construct the watershed attribute data used as explanatory variables and to verify the accuracy of the hydrologic connectivity of the stream reaches, which governs routing and accumulation of mass in the model. Spatial referencing entails the use of GIS techniques to digitally establish the geographic relation between stream reaches in the river network and the various watershed attributes that are used to specify the model. Watershed attribute data sets (e.g., topography, land use, climate) are frequently compiled and reported according to geographic units that are not coincident with the drainage basin boundaries of river reaches; for example at finer spatial scales or according to spatial units that reflect cultural (e.g., counties, states) or other non-hydrologic features of the landscape. Watershed attributes may also be geographically located according to precise locations coded as latitude and longitude, which must be digitally linked to a watershed or nearby river reach. These may include the locations of stream monitoring gages or municipal wastewater treatment outfalls. Verification of automated GIS operations is often required to ensure accurate spatial referencing; including, for example, comparisons of the river name of a gage with that of a river reach or the comparison of the reported drainage area of a gage with that estimated for a river reach. Finally, the SPARROW node and routing architecture also can fully support the modeling of contaminant transport along "off-reach" (landscape) flow paths according to flow directions defined by landscape topography as reflected, for example, in DEMs, although this capability is not currently implemented in the sediment model. Such architecture facilitates the incorporation of high-resolution spatial data sets that delineate sources and other explanatory variables at scales finer than an incremental drainage area for the reach.

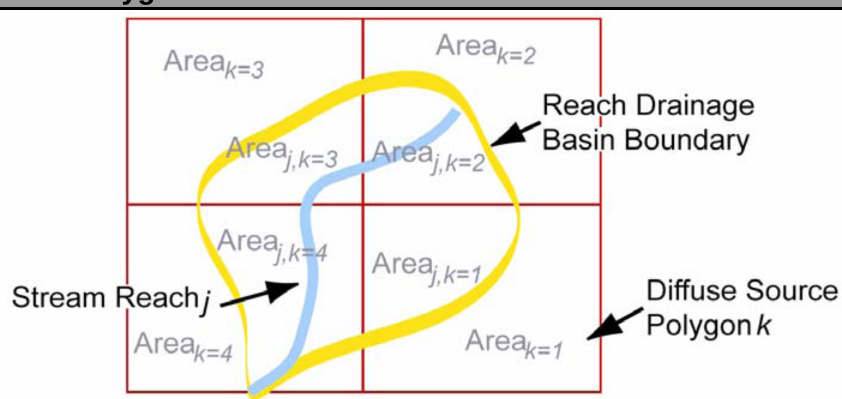
C.2.12 Watershed Sources and Explanatory Variables

The explanatory variables evaluated in SPARROW models reflect current knowledge of natural and human-related sources and the important physical, chemical, or biological properties of the terrestrial and aquatic ecosystems that affect supply and transport of contaminants in watersheds, along with practical considerations of availability. Point- and diffuse-source variables may include direct measures of the introduction or supply of contaminant mass to the landscape and streams and reservoirs (e.g., municipal and industrial wastewater discharge, fertilizer application). Alternatively, source variables may serve as surrogate indicators of the contaminant mass supplied by point and diffuse sources in watersheds, such as land-use/land-cover data or census data on human and livestock populations. Watershed data on sources and other properties (e.g., climate, topography, land-use) must be spatially referenced to the drainage basins and stream reaches of the SPARROW river network. *Figure 12-5* shows the relation between areas of source polygons and the incremental drainage basin of a hypothetical stream reach. Estimates of the diffuse sources associated with the drainage basins of stream reaches can be obtained using GIS operations to digitally overlay the drainage basin boundaries of stream reaches with the polygonal areas associated with the diffuse source data. Once quantitative measures of the overlap in watershed areas are obtained *(Figure 12-5),* estimates of the area-weighted sum of source characteristics ($S_{n,j}$) for reach j and source type n can be calculated as:

$$S_{n,j} = \sum_{k \in P(j)} S_{n,k} \left(A_{j,k} / A^{*}_{k} \right) \qquad \text{(Eq. C-2)}$$

where $P(j)$ is the set of all source-related polygons that intersect the incremental drainage polygon for stream reach j, $S_{n,k}$ is the quantity of source-type n associated with polygon k, $A_{j,k}$ is the sub-area of reach j's incremental drainage that intersects the source-related polygon k, and A^{*}_{k} is the total area of the source-related polygon k.

Figure 12-5: Schematic Illustrating Digital Overlay of Stream Reach Drainage Area and Polygonal Areas Associated with Diffuse Sources



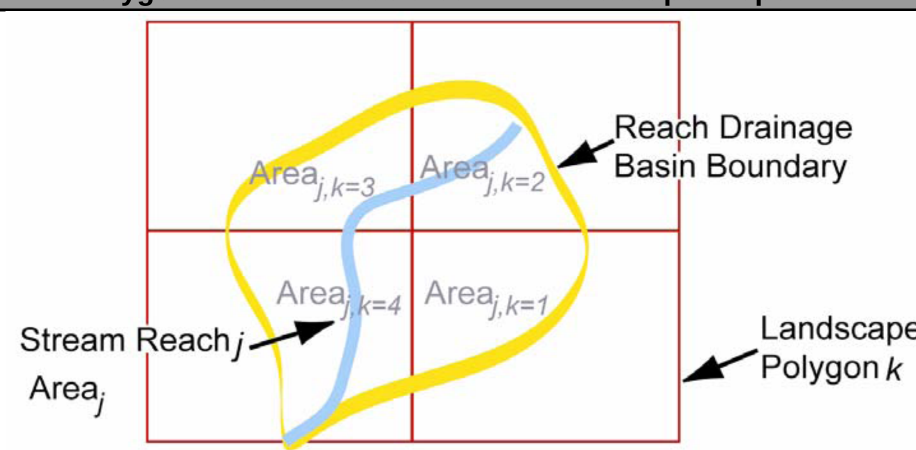
If it is known that a particular contaminant source is associated with a specific land use (or some grouping of land uses) – for example, fertilizer is associated with cultivated land – and if the spatial scale of land-use information is similar to the scale at which watersheds are delineated, then the method described above can be modified to obtain a more refined estimate of sources within a reach incremental drainage area. The enhanced method requires only that the area term $A_{j,k}$ in Equation C-2 be redefined to represent the area of the associated land use that intersects the reach *j* incremental drainage and source polygon *k*, and that the area term A^*_k be redefined to equal the total area of the associated land use within the source polygon *k*.

Climatic and landscape properties that affect contaminant transport may include measures of water-balance terms (e.g., solar radiation, precipitation, evaporation, evapotranspiration), soil characteristics (e.g., permeability, moisture content), water-flow path properties (e.g., slope, hydraulic roughness, topographic index), or management practices and activities (e.g., tile drains, conservation tillage, BMPs). Estimates of various climatic and landscape characteristics or properties are often calculated as an area-weighted mean estimate ($Z_{i,j}$) for stream reach *j* and landscape property *i* according to:

$$Z_{i,j} = \sum_{k \in P(j)} \tilde{Z}_{i,k} \left(A_{j,k} / A^*_j \right) \qquad \text{(Eq. C-3)}$$

where *P(j)* is the set of all land-characteristics polygons that intersect the incremental drainage polygon for stream reach *j*, $Z_{i,k}$ is the landscape property *i* associated with polygon *k*, $A_{j,k}$ is the sub-area of reach *j*'s incremental drainage that intersects the landscape property's polygon *k*, and A^*_j is the total drainage area of the reach watershed *j*. *Figure 12-6* shows the relation between areas of landscape polygons and the incremental drainage basin of a hypothetical stream reach that is described in Equation C-3. Note that the area-weighted mean estimate defined in Equation C-3 differs from the area-weighted sum given by Equation C-2 in that the area ratio terms sum to 1.0 in Equation C-3 but not in Equation C-2.

Figure 12-6: Schematic Illustrating Digital Overlay of Stream Reach Drainage Area and Polygonal Areas Associated with Landscape Properties



The basic SPARROW models described are designed to model long-term mean contaminant loadings in streams, implying explanatory variables in the model should be computed to reflect long-term conditions. Variables that describe contaminant sources and landscape characteristics may be averaged over multiple years, corresponding to the available period of record for water-quality monitoring data, or may reflect the conditions during a specified base year used to estimate stream water-quality loadings, as in the sediment study.

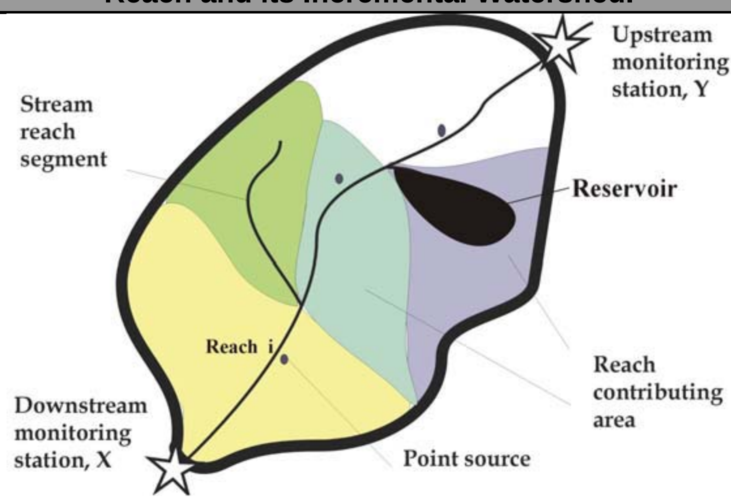
C.3 Model Specification

The specification of a SPARROW model consists of identifying the explanatory variables and functional forms for the associated processes the model is to include. The process is described here in generic terms, supplemented by specific descriptions of the specification of the SPARROW sediment model.

C.3.1 Model Equation and Specification of Terms

Conceptually, the contaminant load or flux leaving a reach is the sum of two components. The first component is the load generated within upstream reaches that is delivered to the reach via the stream network. Losses of flux from the stream network may occur at points where flow is diverted. Additionally, in moving through the reach, flux will generally be attenuated by stream or reservoir processes. The second component consists of source flux that is generated within the reach's incremental watershed and delivered to the stream network somewhere along the reach segment. A number of source-dependent processes, in addition to stream attenuation processes, affect the amount of source flux reaching the stream network and transported to the reach's downstream outlet node. For flux originating on the landscape, the processes affecting delivery to the stream network are called land-to-water delivery processes, and may include both surface and sub-surface elements. A conceptual illustration of the pertinent spatial relations is given in *Figure 12-7.* A connecting reach is generally defined as a stream segment that connects the confluence of two stream segments; a headwater reach is a reach that is defined without an upstream confluence. *Figure 12-7* shows five complete reaches, two of which are headwater reaches, with one of the headwater reaches classified as a reservoir. Each reach is embedded within a

Figure 12-7: Conceptual Illustration of a Reach Network for Five Incremental Watersheds. Model Equation C-4 Describes the Supply and Transport of Load within an Individual Reach and its Incremental Watershed.



Source: McMahon et al. (2003).

color-coded area representing the reach's incremental drainage – the area that drains directly to the reach without passing through another reach. Because there are five reaches, there are five incremental drainage areas. Land-to-water delivery processes determine the amount of contaminant generated within an incremental drainage area, excluding contaminant generated directly on a reach, which is then delivered to the area's corresponding reach. In the figure, two monitoring stations, X and Y, form the boundaries of a nested basin, defined as all reaches above a monitored reach *i*, containing monitoring station X, but exclusive of reaches above and including all upstream monitoring stations – in this case the single monitoring station Y. In-stream attenuation processes associated with headwater reaches affect only the transport of contaminants from their incremental drainage; in-stream attenuation processes for all non-headwater reaches affect the transport of contaminants from their own incremental drainage and also from the incremental drainage of all upstream reaches (including reaches in upstream nested basins).

This conceptual model can be formalized through a mathematical equation. Let F^*_i be the model-estimated flux for contaminant leaving reach *i*. This flux is related to the flux leaving adjacent reaches upstream of reach *i*, denoted by F'_j where *j* indexes the set $J(i)$ of adjacent reaches upstream of reach *i*, plus additional flux that is generated within the incremental reach segment *i*. In most cases, the set of adjacent upstream reaches $J(i)$ will consist of either two reaches, if reach *i* is the result of a confluence, or no reaches if reach *i* is a headwater reach (see *Figure 12-7*). The functional relations determining reach *i* flux are given by:

$$F^*_i = \left[\sum_{j \in J(i)} F'_j\right] \delta_i A\left(Z^S{}_i, Z^R{}_i; \theta_S, \theta_R\right) + \left[\sum_{n=1}^{N_S} S_{n,i} \alpha_n D_n\left(Z^D{}_i; \theta_D\right)\right] A'\left(Z_i^S, Z_i^R; \theta_S, \theta_R\right) \qquad \text{(Eq. C-4)}$$

The first summation term represents the amount of flux that leaves upstream reaches and is delivered downstream to reach *i*, where F'_j equals measured flux, F_j^M, if upstream reach *j* is monitored or, if it is not, is given by the model-estimated flux F^*_j. δ_i is the fraction of upstream flux delivered to reach *i*. If there are no diversions, then δ_i is set to 1. Otherwise, this fraction is defined by the fraction of streamflow leaving upstream reaches that is delivered to reach *i*. $A(.)$ is the stream delivery function representing attenuation processes acting on flux as it travels along the reach pathway. This function defines the fraction of flux entering reach *i* at the upstream node that is delivered to the reach's downstream node. The factor is a function of measured stream and reservoir characteristics, denoted by the vectors Z^S and Z^R, with corresponding coefficient vectors θ_S and θ_R. If reach *i* is a stream, then only the Z^S and θ_S terms determine the value of $A(.)$; conversely, if reach *i* is a reservoir then the terms that determine $A(.)$ consist of Z^R and θ_R.

The second summation term represents the amount of flux introduced to the stream network at reach *i*. This term is composed of the flux originating in specific sources, indexed by $n = 1,\ldots,N_S$. Associated with each source is a source variable, denoted S_n. Depending on the nature of the source, this variable could represent the mass of the source available for transport to streams, or the area of a particular land use. The variable α_n is a source-specific coefficient. This coefficient retains the units that convert source variable units to flux units. The function $D_n(.)$ represents the land-to-water delivery factor. For sources associated with the landscape, this function along with the source-specific coefficient determines the amount of contaminant delivered to streams. The land-to-water delivery factor is a source-specific function of a vector of delivery variables, denoted by Z_i^D, and an associated vector of coefficients $\boldsymbol{\theta_D}$. For point sources that are described by a measured (same units as flux) discharge of mass directly to the stream channel (e.g., municipal wastewater effluent measured in kilograms year^{-1}), the delivery factor should be 1.0, with no underlying factors acting as determinants, and the source-specific coefficient should be close to 1.0. The last term in the equation, the function $A'(.)$, represents the fraction of flux originating in and delivered to reach *i* that is transported to the reach's downstream node. This function is similar in form to the stream delivery factor defined previously; however, the default assumption in SPARROW models is that if reach *i* is classified as a stream (as opposed to a reservoir reach), the contaminants introduced to the reach from its incremental drainage area receive the square root of the reach's full in-stream delivery. This assumption is consistent with the notion that contaminants are introduced to the reach network at the midpoint of reach *i* and thus experience only half of the reach's time of travel. For reaches classified as reservoirs, the default assumption is that the contaminant receives the full attenuation defined for the reach.

The nonlinear model structure in Equation C-4 contains several key features. The additive contaminant source components and multiplicative land and water transport terms are conceptually consistent with the physical mechanisms that explain the supply and movement of contaminants in watersheds. Total modeled flux for a reach is shown to be decomposed into its individual sources. Because this same decomposition is done for all upstream reaches, it is possible in this framework to perform flux accounting, whereby total flux is attributed to its source components. All processes are spatially referenced with respect to the stream network according to the reach in which they operate. This means, for example, that a reservoir at reach *i* affects the transport of all contaminants entering the reach network upstream (but not downstream) of reach *i*. The additive source components also provide a mathematical structure in the model that preserves mass. This can be seen by noting that a doubling of each of the source variables $S_{n,i}$, along with a doubling of all upstream sources, as represented by a doubling of F'_j results in an exact doubling of modeled flux F^*_i. Finally, the modeled flux at any reach *i* is conditioned on monitored fluxes entering the stream network anywhere upstream of reach *i*. This approach to nested basins serves to isolate errors introduced in any upstream basin from incremental errors that arise in a downstream basin, making it defensible to treat nested basins as independent observations.

C.3.2 Contaminant Sources

The selection of potential contaminant source variables in SPARROW models initially depends upon a user's particular knowledge of a watershed as well as inferences that can be derived from the research literature about the major sources that contribute pollutants to watersheds Knowledge of the geography of these sources, based on direct measures or surrogate indicators of the contaminant mass supplied to the land surface and surface waters, is also critical to provide an effective use of the spatially distributed structure in SPARROW. The compilation and availability of large digital spatial data sets has become much more common (e.g., Brakebill and Preston 1999) and the advances in many types of digital topographic and stream network data [e.g., National Hydrography Dataset (NHD), National Land Cover Database (NLCD)] have made spatially distributed modeling more feasible. The inclusion of detailed information in the SPARROW model on the geographic locations where contaminants are released in watersheds has particular relevance to the potential policy and management applications of the model. The source terms used in the model can be generally classified as *intensive* and *extensive* measures of contaminant mass. The former measures are typically descriptive of direct measures of pollutant mass, such as fertilizer application, livestock waste, atmospheric deposition, or sewage-effluent loadings. In these cases, the source-specific parameter (α) is expressed as a dimensionless coefficient that, together with standardized expressions of the land-to-water delivery factor, describes the proportion or fraction of the source input that is delivered to streams. This fraction would be expected to be less than 1.0 but greater than zero, reflecting the removal of contaminants in soils and ground water.

Extensive measures of contaminant mass also may be used in SPARROW models. These are surrogate indicators of contaminant mass and include measures of watershed properties such as specific land-use area and sewered population that are considered to be proportional to the actual mass loadings generated by a general type of a contaminant source. The empirical estimates of the source coefficients in the model provide a quantitative measure of the proportion of the mass loading that is associated with a specified source. If extensive measures are used, the associated model coefficients are expressed as the contaminant mass generated per unit of the source type (e.g., kilograms kilometer^{-2} year^{-1}; kilograms person^{-1} year^{-1}). If combined with the land-to-water delivery factor and expressed as a standardized source coefficient, the coefficient indicates the mean quantity of contaminant mass per unit of the surrogate source measure that is delivered to streams. For land-use terms, the standardized coefficient gives what is frequently cited as an export coefficient (Beaulac and Reckhow 1982; Johnes et al. 1996). Observed values of export coefficients have been reported in the literature for various land-use/land-cover types, such as crop, urban, and forested lands (Beaulac and Reckhow 1982); and often compare favorably with corresponding estimates by SPARROW (e.g., Alexander et al. 2004). Other sources of guidance include sediment erosion estimates generated from the Universal Soil Loss Equation (USLE) as part of the U.S. Department of Agriculture (USDA) National Resources Inventory (NRI) (Natural Resources Conservation Service 2005).

C.3.3 Landscape Variables

Landscape variables in SPARROW describe properties of the landscape that relate to climatic, natural- or human-related terrestrial processes affecting contaminant transport. These include properties that are assumed on some conceptual or empirical basis to control the rates of contaminant processing and transport, and which are widely available. The model structure allows tests of hypotheses about the influence of specific features of the landscape on contaminant transport. Landscape variables may include water-balance terms (e.g., precipitation, evapotranspiration) related to climate and vegetation, soil properties (e.g., organic content, permeability, moisture content), topographic water flow-paths variables (e.g., TOPMODEL overland flow, topographic index, and slope) (Beven and Kirkby 1979), or management practices and activities, including tile drainage, conservation tillage practices, and BMPs related to stream riparian properties. Particular types of land-use classes, such as wetlands or impervious cover, may also be potentially used to describe transport properties of the landscape.

The land-to-water delivery factor in Equation C-4, $D_n(Z_i^D;\theta_D)$, is a source-specific function of a vector of delivery variables, denoted by $\boldsymbol{Z}_i^D$, and an associated vector of coefficients $\boldsymbol{\theta}_D$. In basic SPARROW models, the land-to-water delivery factor has been expressed in exponential functional form. For source n, the fraction of contaminant mass generated in the incremental reach drainage area and delivered to the reach (excluding source coefficient term) is estimated as

$$D_n\left(Z_i{}^D;\theta_D\right)=\exp\left\{\sum_{m=1}^{M_D}\omega_{nm}Z_{m,i}{}^D\theta_{Dm}\right\} \qquad \text{(Eq. C-5)}$$

where $Z_{m,i}{}^D$ represents delivery variable m for the incremental drainage of reach i, θ_{Dm} is its corresponding coefficient, ω_{nm} is an indicator variable that is 1.0 if delivery variable m affects source n and zero otherwise, and M_D is the number of delivery variables. Log-transformed delivery variables, such as the logarithm of soil permeability, may provide an improved fit to the data. Under the log-transformation, the land-to-water coefficient is interpreted as the percent change in flux delivered to streams, derived from all sources to which the land-to-water variable is applied, from a one-percent increase in the land-to-water delivery variable. The full effect of landscape variables on the delivery of source n to streams is determined by the product of the delivery factor in Equation C-5 and the source coefficient α_n. Therefore, modification of delivery factor specification can be expected to change the mean of the delivery factor, resulting in a corresponding counter-adjustment of the source coefficient α_n. In order to improve the interpretability of the source coefficient, and to provide some stability in its value across alternative specifications of the land-to-water delivery factor, it is recommended that the delivery variables $Z_{m,i}{}^D$ in Equation C-5 be expressed as differences from their mean value over all reaches.

C.3.4 Stream Transport

Stream attenuation processes that act on contaminant flux as it travels along stream reaches are frequently modeled as first-order reaction rate processes (Chapra 1997). A first-order decay process implies that the rate of removal of the contaminant from the water column per unit of time is proportional to the concentration or mass that is present in a given volume of water (a zero-order process corresponds to a constant rate of removal per unit of time). In a first-order decay process, the fraction of contaminant removed over a given stream distance is estimated as an exponential function of a first-order reaction rate coefficient (in reciprocal time units) and the cumulative water time of travel over this distance. A reaction rate is estimated on a volumetric basis, and therefore is expected to depend on properties of the water column that are proportional to water volume, such as streamflow and water-column depth (Stream Solute Workshop 1990). Accordingly, in basic forms of the SPARROW model, the fraction of the contaminant mass originating from the upstream node and transported along reach i to its downstream node is estimated as a function of the mean water time of travel ($T_{c\ i}^{S}$; units of time) in reach i and stream class c defined according to discrete intervals of mean streamflow or depth (in this case, $Z_i^S = \{T_{c\ i}^{S}\}$, $c = 1,\ldots,C_S$, where $T_{c\ i}^{S}$ is nonzero only for the streamflow class corresponding to reach i), and a stream-size dependent loss rate coefficient (θ_{Sc}; units of time) such that:

$$A\left(Z_i^{\ S}, Z_i^{\ R}; \theta_S, \theta_R\right) = \exp\left\{-\sum_{c=1}^{C_S} \theta_{Sc} T_c^{\ S}{}_i\right\} \qquad \text{(Eq. C-6)}$$

Mean water time of travel is estimated as the quotient of the reach channel length and mean water velocity. The most accurate estimates of water velocity are obtained from time-of-travel dye studies (Jobson 1996), but in some cases may be obtained from instantaneous measurements of water velocity taken during flow gage site visits. Empirical geomorphic relations, which use regression methods to relate time-of-travel measurements to channel and basin properties (e.g., streamflow, slope), also are available for regions of the U.S. and other countries (e.g., Jobson 1996; USEPA 1996b; Jowett 1998; Alexander et al. 1999) and can be used to estimate the time-of-travel of stream reaches for a given river network. Alternatively, channel length may be used in Equation C-6 where water velocity estimates are not available. This assumes that the water time of travel is proportional to the channel length, and estimates of the removal rate are expressed as reciprocal length (e.g., see Alexander et al. 2002a; McMahon et al. 2003).

C.3.5 Reservoir and Lake Transport

Attenuation processes that act on contaminant mass as it travels through a lake or reservoir are often modeled as a net removal process, with the loss coefficient expressed as either a first-order *reaction rate* or a *mass-transfer coefficient* (also referred to as an *apparent settling velocity*) (Chapra 1997). Both of these loss expressions have been used in empirical mass-balance lake models for phosphorus (e.g., Vollenweider 1976; Reckhow and Chapra 1983) and nitrogen (e.g., Kelly et al. 1987; Molot and Dillon 1993), although the use of the mass-transfer rate is more common. These mass-balance models typically assume steady-state and uniformly mixed conditions in the waterbody. The *reaction rate* is expressed in reciprocal time units and its estimation and use are dependent on knowledge of the water depth and surface area of the waterbody (volume). The *apparent settling velocity* is expressed in units of length per time and is estimated as a function of the ratio of the reservoir outflow and the surface area of the reservoir sediments (assumed equal to the surface area of the waterbody); this ratio, denoted q_i^R for reach i, is termed the areal hydraulic load and is a measure of the water displacement or velocity in the reservoir. The term "apparent" indicates that the settling velocity measures the net effect of various processes that remove the contaminant from the water column and deliver it to the sediments, and processes that may add contaminant to the water column. The function that relates the areal hydraulic load to the fraction of flux attenuated in a reservoir is

$$A\left(\mathbf{Z}_i^S, \mathbf{Z}_i^R; \boldsymbol{\theta}_S, \boldsymbol{\theta}_R\right) = \frac{1}{1 + \theta_{R0}\left(q_i^R\right)^{-1}} \qquad \text{(Eq. C-7)}$$

where θ_{R0} is a parameter, to be estimated in the model, representing the apparent settling velocity.

C.4 Model Estimation

The SPARROW model equation, given in Equation C-4, is a nonlinear function of its parameters, and the model must be estimated using nonlinear techniques. The model errors are assumed to be independent across observations with zero mean, while the variance of each observation may be observation specific. SPARROW utilizes a nonlinear weighted least squares (NWLS) estimation method, which does not assume the precise distribution of the residuals. Several algorithms exist for obtaining NWLS estimates, and the Levenberg-Marquardt Least-Squares method, implemented in SAS, is used to estimate the SPARROW model (SAS 1999). Unlike linear models, the statistical properties of the estimated parameters for nonlinear models are not precisely known in finite samples. For this reason, much of the theory of nonlinear estimation has focused on characterizing asymptotic properties – the statistical properties of the coefficient estimates as the sample size goes to infinity. Emphasis in the following sections is on interpretation of estimated model parameters.

C.4.1 Evaluation of Model Parameters

The objective of parameter evaluation in SPARROW modeling is to determine whether a converged model provides *statistically sound* and *physically interpretable* coefficient values. The first objective entails the appraisal of model parameters for statistical significance and the quantification of uncertainty. This provides important information for identifying unique model specifications (parameters and values for which the model predictions are sensitive) and determining the level of model complexity (number and types of explanatory variables and model functions) that can be empirically supported by the stream monitoring data. The emphasis on parameter estimation in SPARROW models reflects the objective of identifying the important contaminant sources and factors affecting mean-annual contaminant transport over large spatial scales in soils and in ground and surface waters. The key parameter statistics include the estimated mean coefficient values, estimated variance of these coefficient estimators, and measures of statistical significance based on the *t* statistics (ratio of the coefficient value to its standard error). These statistics are biased in finite samples but consistent as sample size goes to infinity; the *t* statistics are asymptotically distributed as standard normal. The *p*-values are based on a two-tailed probability from a Student's *t* distribution. The *p*-values can be used to identify statistically significant model coefficients – those that are statistically distinguishable from zero – and can be used to refine the parameter set to identify parsimonious SPARROW models.

The second complementary objective is the evaluation of the parameters for physical interpretability. This objective entails the evaluation of the sign and magnitude of model coefficients to test hypotheses about the importance of different contaminant sources and the hydrologic and biogeochemical processes that are represented by the explanatory variables of the model. SPARROW model parameters reflect the net effects over large spatial scales of an aggregate set of hydrologic and biogeochemical processes and human-related activities. The interpretability of the parameters and their relation to specific processes is enhanced in SPARROW by the use of a mass balance, mechanistic structure that explicitly separates the terrestrial and aquatic properties of watersheds and accounts for nonlinear interactions among watershed properties, together with an emphasis on the statistical estimation of parameter values. The sign of SPARROW model coefficients can be evaluated to determine the direction of influence of any explanatory variable on in-stream estimates of mean-annual flux. The direction of influence is assessed for consistency with the anticipated response based on available theoretical or empirical information about processes related to individual explanatory factors. For example, a negative sign on the soil permeability coefficient indicates that total loads in streams are inversely related to permeability – i.e., in-stream loads of contaminants are generally lower in watersheds with highly permeable soils. The sign of the coefficient is also important in estimating physically meaningful contaminant source terms in SPARROW, which are generally expected to contribute positive contaminant mass to the watershed system.

Coefficients associated with source inputs expressed in areal units describe the mass per unit area delivered to streams from these land areas. These areal expressions of contaminant transport (export) can be directly compared with ranges of export coefficients reported in the literature (e.g., Beaulac and Reckhow 1982). Estimated SPARROW coefficients associated with specific land uses generally compare favorably with published export coefficients. Other source coefficients that are expressed in dimensionless units provide a measure of the fraction of the contaminant that is delivered from each source to streams, rivers, and reservoirs. These coefficients can be evaluated to determine how reasonably they reflect the net mean rates of contaminant removal by a source as part of the delivery to aquatic systems.

C.5 Model Prediction

SPARROW output contains prediction results paired with measures of accuracy. A number of technical issues can arise in the derivation of these statistics; most of these caused by the nonlinear nature of the SPARROW model. The prediction equation is similar to the calibration Equation C-4. Because predictions are generated for specific sources, however, it is necessary to decompose flux into source-specific components. Let $F^{*}_{i,n}$ denote the reach i model-estimated flux associated with source n, and let

$F'_{j,n}$ be the source-n flux from upstream reach j. If reach j is monitored (that is, $j \in I$, where I is the set of

monitored reaches), and predictions are conditioned on measured flux, then $F'_{j,n}$ is apportioned from the measured flux F_j^M according to

$$F'_{j,n} = \frac{F^{*}_{j,n}}{\sum_{m=1}^{N_y} F^{*}_{j,m}} \cdot F_j^{M} \qquad \text{(Eq. C-8)}$$

Otherwise, if reach j is not a monitored reach (that is, $j \notin I$) or predictions are not conditioned on

measurements, then $F'_{j,n}$ is set equal to $F^*_{j,n}$. The equation defining the model-estimated flux for source n, $F^*_{i,n}$, is given by

$$F^*_{i,n} = \left[\sum_{j \in J(i)} F'_{j,n} \right] \delta_i A\left(Z^S{}_i, Z^R{}_i; \theta_S, \theta_R\right) + S_{n,i} \alpha_n D_n \left(Z_i{}^D; \theta_D\right) A'\left(Z_i{}^S, Z_i{}^R; \theta_S, \theta_R\right) \quad \text{(Eq. C-9)}$$

Appendix D – A Preliminary SPARROW Model of Suspended Sediment for the Coterminous United States (Schwarz 2008a)

A Preliminary SPARROW Model of Suspended Sediment for the Conterminous United States

Open-File Report 2008–1205

U.S. Department of the Interior
U.S. Geological Survey

A Preliminary SPARROW Model of Suspended Sediment for the Conterminous United States

By Gregory E. Schwarz

National Water-Quality Assessment Program

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Contents

Figures

Tables

A Preliminary SPARROW Model of Suspended Sediment for the Conterminous United States

By Gregory E. Schwarz

Abstract

This report describes the results of a preliminary Spatially Referenced Regression on Watershed attributes (SPARROW) model of suspended sediment for the conterminous United States. The analysis is based on flux estimates compiled from more than 1,800 long-term monitoring stations operated by the U.S. Geological Survey (USGS) during the period 1975-2007. The SPARROW model is structured on the Reach File 1 (RF1) stream network, consisting of approximately 62,000 reach segments. The reach network has been modified to include more than 4,000 reservoirs, an important landscape feature affecting the delivery of suspended sediment. The model identifies six sources of sediment, including the stream channel and five classes of land use: urban, forested, Federal nonforested, agricultural and other, noninundated land. The delivery of sediment from landform sources to RF1 streams is mediated by soil permeability, erodibility, slope, and rainfall; streamflow is found to affect the amount of sediment mobilized from the stream channel. The results show agricultural land and the stream channel to be major sources of sediment flux. Per unit area, Federal nonforested and urban lands are the largest landform sediment sources. Reservoirs are identified as major sites for sediment attenuation. This report includes a description for how the model results can be used to assess changes in instream sediment flux and concentration resulting from proposed changes in the regulation of sediment discharge from construction sites.

Introduction

This report describes the results of a preliminary Spatially Referenced Regression on Watershed attributes (SPARROW) model of suspended sediment for the conterminous United States.

Sediment Model Data Sources

The spatial framework of the SPARROW sediment model is the vector-based 1:500,000-scale River Reach File (RF) 1 hydrography, originally developed by the U.S. EPA (U.S. Environmental Protection Agency, 1996) and subsequently enhanced to include areal hydraulic load information for selected reservoirs (Ruddy and Hitt, 1990), shoreline reaches, and reach catchment areas derived from the U.S. Geological Survey (USGS) HYDRO1k Digital Elevation Model (DEM) (U.S. Geological Survey, 2006a). The enhanced network (Nolan and others, 2002), consisting of 62,776 reach segments, including shoreline reaches, 61,214 delineated reach catchments, and 2,171 individual reservoirs, has been used to support numerous national SPARROW modeling efforts for the conterminous United States (for example, Alexander and others, 2000). The RF1 reach network for the current SPARROW sediment model was further enhanced by the inclusion of areal hydraulic load information for approximately 2,000 additional large reservoirs (reservoir storage greater than 500 acre-feet), identified from the National Inventory of Dams (NID) (U.S. Army Corps of Engineers, 2006) and linked to the RF1 network according to dam geographic coordinates, river name, and drainage area.

Included in the original RF1 network are reach estimates of mean streamflow and mean velocity, the latter being converted to reach time of travel using the RF1 measure of reach length. The attributes of mean streamflow and mean velocity are used to assess various sediment mobilization and attenuation processes associated with the stream channel. Because catchment areas used to derive the original RF1 estimates of mean streamflow are not compatible with catchments included in the enhanced RF1 network, an alternative measure of mean streamflow was used to compute flow-weighted sediment concentration. The alternative measure is based on an interpolation of USGS streamgage estimates from the 1975-2006 water years (WYs), with extrapolation of streamflow upstream of gages based on runoff estimated at downstream or neighboring stations and apportioned to the land surface according to the enhanced RF1 catchments (David Wolock, U.S. Geological Survey, 2008, written commun.). The available period for these data is 1 year less than the period for this study, and a water year consists of the period October 1st of the previous calendar year through September 30th of the enumerated water year.

The dependent variable in the SPARROW sediment model is given by long-term mean sediment flux. Long-term mean sediment flux is estimated using the maximum likelihood approach developed by Cohn (2005), as implemented in the Fluxmaster program (Schwarz and others, 2006). Instream sediment concentrations and stream discharge measurements over the WY period 1975-2007 have been obtained from the National Stream Quality Accounting Network (NASQAN) (Alexander and others, 1996; U.S. Geological Survey, 2006b), the USGS National Water-Quality Assessment (NAWQA) Program (Mueller and Spahr, 2005), and the USGS National Water Information System (NWIS) (U.S. Geological Survey, 2008), a database encompassing USGS water-quality monitoring stations as well as water-quality monitoring activities done in cooperation with State governments. Sampling for suspended sediment (USGS water-quality parameter 80154) is typically done periodically, not daily. Sample data are weighted by channel cross-sectional flow geometry (depth, width) and correlated with stream discharge at the time of sampling. A linear regression model is estimated that relates log-transformed instantaneous suspended-sediment concentration to log-transformed mean daily streamflow, which is measured continuously via water-surface height (stage) coupled with a previously estimated relation between surface height and instantaneous flow. Included in the regression are the sine and cosine of decimal time to capture a seasonal signal, and a linear time trend to be used for detrending flux. To support the detrending of flux, a companion model of daily streamflow is estimated for each water-quality station. The streamflow model relates the logarithm of daily streamflow to a second-order harmonic of the sine and cosine of decimal time, and a linear time trend term. To account for serial correlation in the daily values, the model is estimated using time-series methods that assume a 30-day autoregression in the residuals. The water-quality and streamflow models are used to simulate daily flux, with both water-quality and streamflow trends removed, for all days within the 33-year period WY 1975-2007 for which a daily streamflow value is available for every day in the same WY. Thus, if streamflow is not available for any day within a given WY, no simulated water-quality flux is computed for any day in that WY. The simulated estimates of flux, detrended to the base year 1992, for all complete WYs within the 33-year period, are averaged to obtain a detrended flux reflecting long-term mean hydrologic conditions.

The sediment model is estimated using 1,828 monitoring stations located on the RF1 stream network (fig. 1). Stations were selected for inclusion in the model if they had at least 15 concentration measurements during the period WY 1975-2007, and the standard error of the flux estimate did not exceed 80 percent of the flux estimate. Of the stations included in the analysis, 90 percent had streamflow records in excess of 5 years, 70 percent had records exceeding 19 years, and 50 percent had records that exceeded 32 years. Approximately 700 monitoring stations have been indexed to the RF1 stream

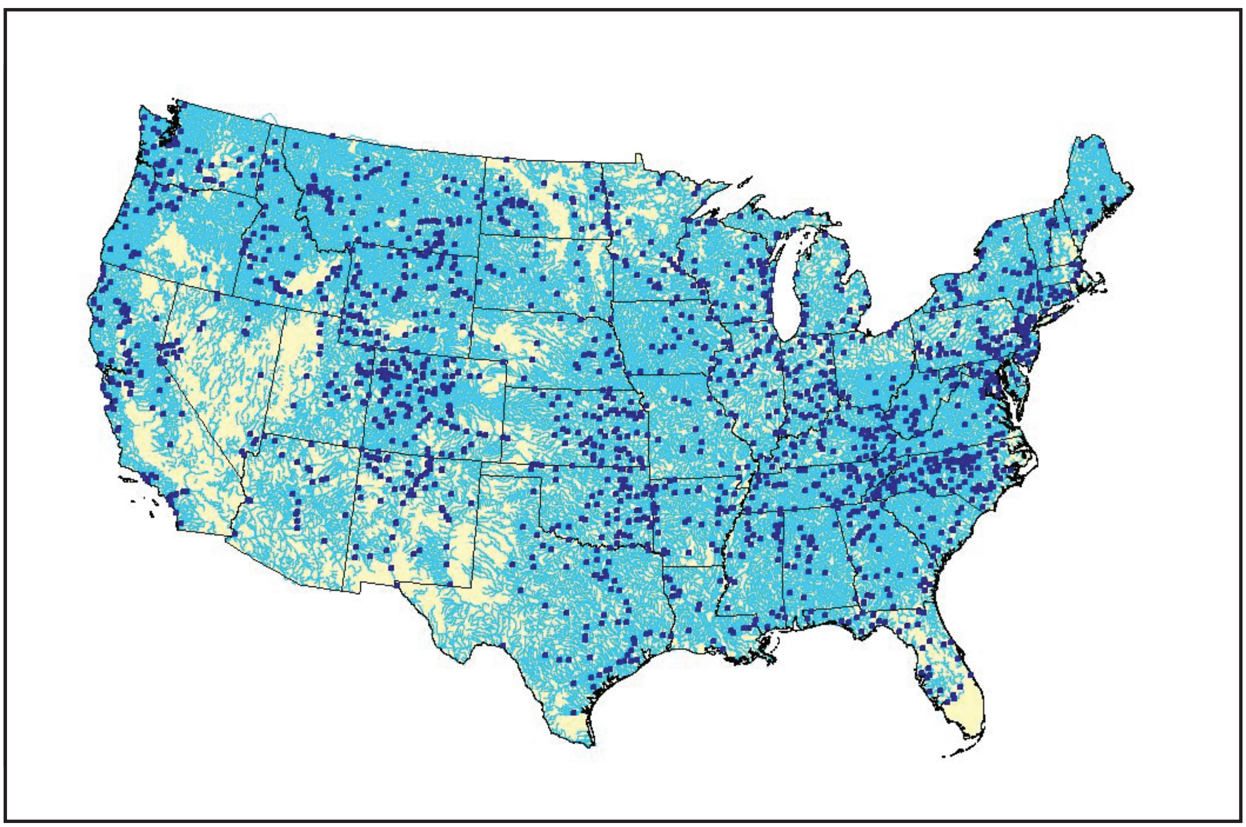

Figure 1. Location of 1,828 water-quality monitoring stations used in the SPARROW sediment model, in relation to the Reach File 1 (RF1) stream network.

network as part of previous studies (Alexander and others, 2000). The remaining monitoring stations were linked to RF1 reaches via their association with USGS streamgages, which are linked to the National Hydrography Dataset (NHD) reach network (Stewart and others, 2006). The location of these streamgages was transferred from NHD to RF1 using information on streamgage latitude and longitude (based on the NHD location), stream name, and reported drainage area. If multiple stations were present for the same RF1 reach, the alternative stations were first ranked in terms of drainage area, number of days predictions were made, the number of water-quality observations used in estimation, the coefficient of variation of the flux estimate, and whether the water-quality and streamflow records were sufficient to support detrending of flux to the base year. (To be detrended, the water-quality and streamflow records must span at least 3 years and, if extended no more than 15 percent in duration, must include the detrending date June 30, 1992.) The station with the lowest sum of these ranks was selected for inclusion in the model.

Most of the source variables for the SPARROW sediment model are expressed as extensive measures of land use. Data on land cover and land use have been developed from the 2001 USGS National Land Cover Data (NLCD) Set Retrofit Change Product [Multi-resolution Land Characteristics (MRLC), 2001], derived from Landsat Thematic Mapper/Embedded Trace Macrocells (TM/ETM) remotely sensed imagery at 30-meter resolution and classified according to the eight Anderson Level I categories. These data were transformed to 1-square kilometer (km^2) cells within a Lambert map projection as consistent with HYDRO1k for use within SPARROW. The 1-km^2 cells are then resolved to catchments associated with specific RF1 reaches. For model estimation, land use was assigned the 1992 values of the 2001 NLCD Retrofit Change Product; the 2001 values of the Retrofit Change Product were used to simulate water-quality conditions for 2001.

Federal nonforested (range and barren) land was included in the model separately from private land. Federal land extent, taken from the Federal Land coverage of the National Atlas (U.S. Geological Survey, 2003) and transformed to 1-km^2 cells in Lambert projection, was apportioned into Federal range and Federal barren land using the 1-km^2 transformation of the 1992 NLCD Change Product for Anderson Level I range and barren land classes (see above). For model simulation of 2001 conditions, Federal range and barren land were similarly estimated using the land-use estimates from the 2001 NLCD Retrofit Change Product.

Variables governing the estimated delivery of contaminants from the land to RF1 streams include soil erodibility [the revised universal soil loss equation (RUSLE) K factor], soil permeability (inches/hour; depth integrated), mean slope (percent), and precipitation (RUSLE R factor). Slope, soil erodibility, and permeability were obtained from the State Soil Survey Geographic (STATSGO) database (U.S. Department of Agriculture, 1994), converted to a 1-km^2 grid in the Lambert projection, and averaged over RF1 catchments. The RUSLE rainfall factor was derived by interpolating a digitized national map of rainfall factor isoline contours (Wischmeier and Smith, 1978), creating a continuous 1-km^2 grid surface in the Lambert projection. The grid coverage was subsequently averaged over individual RF1 catchments.

Preliminary Model Estimation Results

The nonlinear, least-squares estimation results of a preliminary version of the SPARROW suspended sediment model are given in table 1. The preliminary model includes six source terms, five of which are measured by area of specific land use (urban, forested, Federal nonforested, agricultural, and other land, expressed in km^2), and an additional source given by the length of the stream channel. The Federal land class consists only of Federal range and barren land; it excludes Federal forested land, which is incorporated in the forested land class. Agricultural land includes cropland, pasture land, and orchards. Other land consists of non-Federal range and barren land. Among all the land classes, only wetlands and land covered by water, ice, or snow are excluded as a potential source. The source described as “streambed” relates to stream channels as a direct source of sediment, and is measured in terms of stream length (expressed in meters).

The transport of sediment from the land surface to RF1 rivers is mediated by a land-to-water delivery factor that is expressed as a function (see Equation 1.28 in Schwarz and others, 2006) of logarithm-transformed values of soil permeability, soil erodibility (USLE K factor), land-surface slope, and the USLE rainfall factor. The streambed source is mediated by the logarithm of streamflow, distinguished by streamflows above and below 500 cubic feet per second (ft^3/s). Although the mediation of the streambed source by streamflow is not a land process, the manner in which the process is specified in SPARROW is mathematically equivalent to treating streamflow as a land-to-water variable affecting the streambed source, and for this reason streamflow is listed as a land-to-water variable in table 1. To facilitate interpretation of the source coefficients, the delivery variables are all expressed as deviations from their mean value.

Streamflow was used as the mediating factor affecting the mobilization of sediment from the stream channel. There are two principal physical factors affecting the mobilization of sediment from the streambed: the energy of the stream, represented by stream velocity; and the availability of channel material, which is proportional to the area of the streambed per unit of channel length. Direct measurements of channel velocity, width, and depth (the determinants of streambed area per unit channel length) are not available for most reaches in the RF1 network. Streamflow was deemed to be a viable surrogate for these variables because it is highly correlated with them (Schwarz and others, 2006) (in fact, the estimate of velocity included with the RF1 network is derived from streamflow), and because estimates of streamflow exist for all reaches.

The model specifies two instream sediment-attenuation processes: attenuation in streams (see Equation 1.30 in Schwarz and others, 2006), distinguished by three streamflow classes (less than 500 ft^3/s, 500-1,000 ft^3/s, and greater than 1,000 ft^3/s); and reservoir attenuation, specified as a function of areal hydraulic load (see Equation 1.34 in Schwarz and others, 2006). The three streamflow classes used to distinguish instream decay are characteristic of the streamflow at the 1,828 monitoring stations: 1,040 stations have streamflow less than 500 ft^3/s, and 539 stations have streamflow exceeding 1,000 ft^3/s. Attenuation in reservoirs is specified to be a function of the ratio of the reservoir settling velocity, the estimated mean rate at which sediment moves vertically in water, to areal hydraulic load (the ratio of streamflow to reservoir surface area), which represents the velocity a particle at the surface of the stream would need to travel in order to reach the bottom of the reservoir within the average period that the streamflow is impounded in the reservoir.

The estimation results given in table 1 characteristically reflect the large uncertainty associated with sediment modeling. The model root mean squared error (RMSE) is 1.4, implying that predicted sediment flux or concentration in any given reach has an error of approximately 140 percent (Schwarz and others, 2006). This compares with the much smaller 0.3 RMSE obtained with total nitrogen models (Alexander and others, 2008). Despite this uncertainty, many of the model coefficients are statistically significant. With the exception of forested land, all of the source variables are highly statistically significant. The largest intrinsic sediment yield is associated with Federal range and barren land; urban land has the second highest intrinsic yield. Stream channels are also a statistically significant source of sediment.

Land-to-water delivery for land sources is strongly mediated by the four delivery variables—soil permeability, soil erodibility, slope, and rainfall. As would be expected and with the exception of soil permeability, the presence of higher levels of these factors results in greater sediment delivery to streams. Conversely, permeable soils reduce the delivery of sediment, presumably because more of the water runoff infiltrates into the ground leaving less overland flow to transport

Table 1. Preliminary estimation results for the SPARROW suspended sediment model.

[$Kg/km^2/yr$ = kilograms per square kilometer per year; kg/m/yr = kilograms per meter per year; ft^3/s = cubic feet per second; and m/yr = meters per year]

Parameter	Units	Estimate	Standard Error	p-value
Source Coefficients				
Urban land	$kg/km^2/yr$	47,130	9,925	0.000
Forested land	$kg/km^2/yr$	634	898	0.480
Federal non-forested land	$kg/km^2/yr$	64,344	12,411	0.000
Agricultural land	$kg/km^2/yr$	18,047	3,623	0.000
Other land	$kg/km^2/yr$	11,343	3,186	0.000
Streambed (reach length)	kg/m/yr	28.80	6.40	0.000
Land-to-Water Delivery Factors				
Slope	–	0.804	0.087	0.000
Soil permeability	–	-0.778	0.094	0.000
R-factor	–	0.821	0.081	0.000
K-factor	–	1.292	0.279	0.000
Flow [< 500 ft^3/s] (Reach)	–	0.154	0.100	0.125
Flow [> 500 ft^3/s] (Reach)	–	0.721	0.354	0.042
Stream Attenuation Factors				
Travel time ($Q < 500$ ft^3/s)	day^{-1}	-0.007	0.016	0.673
Travel time ($500 < Q < 1{,}000$ ft^3/s)	day^{-1}	-0.233	0.057	0.000
Travel time ($Q > 1{,}000$ ft^3/s)	day^{-1}	0.009	0.047	0.854
Reservoir settling velocity	m/yr	36.49	5.552	0.000
Number of Observations	1,828			
Root mean squared error (RMSE)	1.414			
R-square	0.711			

sediment. Greater streamflow causes an increase in the amount of sediment generated from stream channels, with the largest effect associated with streams having flows greater than 500 ft^3/s.

Reservoir retention is statistically significant and indicates sediment settles at a mean velocity of 36 meters per year (m/yr), comparable to estimates of reservoir attenuation obtained for phosphorus (Alexander and others, 2008). The preliminary model indicates medium-sized streams (flow 500-1,000 ft^3/s) have a statistically significant negative rate of instream attenuation, indicating that medium streams are a source of sediment, in addition to the stream-channel source identified above. Unlike the streambed source of sediment (see above), which is dictated by channel length and, thus, explicitly tied to a physical entity, the implied "source" of sediment arising from a negative rate of instream attenuation represents a proportional "enhancement" of sediment already suspended in the stream (see Equation 1.30 in Schwarz and others, 2006). Because this proportional enhancement of sediment is not associated with a physical source, it is inconsistent with mass balance and represents an anomalous finding of the model that is not yet explainable. Instream attenuation in small and large streams is not significantly statistically different from zero. Thus, the preliminary model does not find evidence for sediment loss in streams.

Model Simulation

The estimated SPARROW suspended sediment model for base WY 1992 can be used to simulate water-quality conditions for 2001, with and without U.S. EPA proposed changes in the regulation of construction activity. The simulation of suspended sediment flux for WY 2001, without changes in regulation, is obtained using the model prediction equation, described as Equation 1.120 in Schwarz and others (2006), with all land use-related source variables set to 2001 values according to the 2001 NLCD Retrofit Change Product. Flow-weighted average sediment concentration is estimated by dividing simulated flux estimates by mean streamflow over the period WY 1975-2006, obtained from USGS streamgages and interpolated to RF1 reaches (Wolock, U.S. Geological Survey, 2008, written commun.). Although the SPARROW model does not explicitly include a source term for construction, such loading is implicitly accounted for in the urban land component of the model. Therefore, the 2001 precompliance loading from construction (that is, the "base-case" scenario loading) is incorporated in the 2001 loading attributed to urban land that is obtained by evaluating the urban land variable in the SPARROW model using the 2001 NLCD Change Product value.

The absence of an explicit term for construction loading in the SPARROW model necessitates the development of an indirect method for assessing changes in sediment loading arising from different construction industry regulation scenarios. Suspended sediment loading under alternative regulation scenarios has been estimated by U.S. EPA using a variation of the Universal Soil Loss Equation (USLE). The USLE method determines the amount of soil that is mobilized and delivered, under a proposed regulation scenario, to the edge of a construction site. To evaluate the impact that changes in these loadings have on RF1 stream-sediment flux and flow-weighted concentration, it is first necessary to assess the rate at which "edge of site" loads are subsequently delivered to RF1 streams. To do this, we use the estimated rate of delivery from agricultural land, a source that is explicitly included in the model, and that can be factored into a mobilization, "edge of site" delivery component and a stream-delivery component. The method described below isolates the stream-delivery component from the overall rate of delivery from agricultural land and applies this component to the change in construction loading to determine the change in loading to RF1 streams. Thus, the approach assumes that the delivery of sediment from the edge of a site to an RF1 stream is the same for both construction and agriculture activities; the mobilization and delivery of sediment to the edge of the site between these activities is allowed to differ. Given that urban areas, as compared to agricultural areas, generally exhibit higher rates of runoff, with compressed runoff duration periods for a given precipitation event, the assumption probably leads to an underestimate in the change in stream-sediment flux from proposed regulation of the construction industry. It would not be necessary to make this assumption if the analysis was based instead on a factorization of urban land delivery; however, as is indicated below, the information necessary to do this is not available.

The amount of sediment mobilized from agricultural land, delivered to the edge of field, and subsequently transported to an RF1 stream, is estimated in SPARROW as the product of the agricultural land-source coefficient and the associated land-to-water delivery factor. This quantity, denoted K_{AG} and expressed in units of yield as kilograms per square kilometer per year ($kg/km^2/yr$), is conceptually divided into two components: a component representing the amount of sediment mobilized from agricultural land and delivered to the edge of site, denoted $K_{EOS\text{-}AG}$, and a component representing the fraction of this material that is subsequently delivered to an RF1 stream, denoted K_{RF1}. If $\Delta L_{EOS}(S)$ represents the change in construction sediment loading to the edge of site, as estimated by U.S. EPA using the USLE method, and $\Delta L_{RF1}(S)$ represents the change in sediment loading to RF1 streams associated with construction regulation scenario S, then the two loadings are related according to

$$\Delta L_{\mathrm{RF1}}(\mathrm{S}) = K_{\mathrm{RF1}} \cdot \Delta L_{\mathrm{EOS}}(\mathrm{S}) = K_{\mathrm{AG}} \frac{\Delta L_{\mathrm{EOS}}(\mathrm{S})}{K_{\mathrm{EOS\text{-}AG}}} \tag{1}$$

Given estimates of the regulation-induced change in sediment delivered to each RF1 reach, instream processes associated with attenuation in channels and reservoirs, as described by the SPARROW model estimates, can be applied to estimate changes in sediment flux and concentration for all RF1 reaches. Additionally, because SPARROW imposes mass balance, the reservoir attenuation process can be used to assess changes in the amount of sediment retained in each of the approximately 4,000 reservoirs linked directly to the RF1 network.

To implement the method described by Equation 1, it is necessary to have estimates of K_{AG}, the delivery factor for agricultural land, and $K_{EOS\text{-}AG}$, the amount of soil erosion mobilized from agricultural land and delivered to the edge of site. K_{AG} is reach specific and is estimated in SPARROW as a function of the land-to-water delivery factors for agriculture (see Equation 1.28 in Schwarz and others, 2006) and the empirically estimated values of the agricultural land-source and land-to-water delivery coefficients. The value for $K_{EOS\text{-}AG}$ is obtained using information on soil erosion included in the 1992 National Resources Inventory (NRI) (U.S. Department of Agriculture, 1994), which reports county estimates of soil erosion rates for cropland, pasture, and orchards—the land classes encompassed by the class labeled "agricultural land" in the NLCD Retrofit Change Product. (K_{RF1} is based on agricultural land because erosion rates for urban land are not reported by the NRI.) The USLE-based county erosion rates for each of the three land classes were weighted according to the share of county land in the respective class (as reported in the 1992 NRI) and then averaged. If there was no county estimate for cropland erosion, an average erosion rate for that county was not computed. The county average erosion rate was apportioned to RF1 catchments according to a 1-km^2 grid of agricultural land area derived from the 1992 NLCD Retrofit Change Product. For 6,329 catchments where an NRI erosion rate was not available, the erosion rate was estimated by determining an agricultural land weighted average of all available erosion rates for catchments in the same 8-digit hydrologic cataloging unit. Lack of an 8-digit cataloging unit average necessitated using a 6-digit cataloging unit average for 713 catchments, and the remaining 54 catchments were estimated using a 4-digit cataloging unit average.

Model Limitations

The preliminary SPARROW model for suspended sediment described above has certain limitations, some of which are inherent to the methodology and some the result of the particular model application. An example of the former is the restriction of the analysis to the description of long-term mean water-quality conditions. As explained in Schwarz and others (2006), this restriction is a consequence of imposing mass balance on the predictions. One of the benefits of the mass balance methodology is that it facilitates the interpretation of model coefficients, and enables the comparison of coefficient estimates to estimates obtained by other studies in the literature; however, the restriction to mean water-quality conditions precludes an analysis of the frequency with which conditions of extreme sediment transport occur.

A second example of a methodology-imposed limitation concerns the use of the statistical method to estimate model coefficients. The statistical method provides considerable insight into the evaluation of model fit and the empirical relevance of individual model processes. It also enables the estimation of prediction uncertainty. However, reasonable precision in the statistical estimation of model coefficients is generally possible only if the number of specified model parameters is limited to those associated with sources and delivery processes that have the greatest influence on water quality. The resulting model is parsimonious, but may be overly simplistic in terms of the range of processes affecting sediment transport.

The RF1 reach network is fairly coarse, and its use in the present application limits the ability to predict water-quality conditions in smaller streams. The median headwater catchment area in RF1 is 88 km^2, implying water-quality conditions in streams with smaller catchments are unresolved. Additionally, the smallest monitored catchment in the SPARROW sediment model is 13 km^2, with only five percent of the monitored catchments less than 100 km^2. A SPARROW analysis structured on a denser reach network, such as the National Hydrography Dataset, would relax the reach network limitation, but the large number of reaches associated with this network would make it difficult to conduct a national analysis.

The large error obtained in the present analysis implies the prediction of sediment flux or concentration in any given reach segment is imprecise. Although this error compromises the ability to describe water-quality conditions in any given reach, it does not preclude using the model to characterize water quality in a large grouping of reaches. As long as the error across reaches is sufficiently independent, the assessment of mean water quality in a group of reaches becomes more precise as the size of the group increases.

With the exception of reservoirs, the preliminary model does not find evidence of sediment attenuation in streams. The result implies that sediment transport in streams is not in a steady state. Additional investigation is necessary to determine if this result is real, or if there are additional reach attributes, currently absent from the model, that identify a subset of reaches where sediment attenuation takes place.

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Appendix E – Estuary TSS Concentration Calculations Using Dissolved Concentration Potentials

As described in *Section 6.3*, EPA used SPARROW predicted sediment loadings in combination with dissolved concentration potentials (DCPs) for calculating sediment concentrations where flow estimates were not available from SPARROW. This appendix includes all the tables referenced in *Section 6.3*.

DCP values and baseline TSS concentrations calculated using DCPs are shown in *Table E-1* through *Table E-4* for Southeastern, Gulf, Northeastern, and West Coast estuaries, respectively. Data on these and additional estuaries are found via the NOAA Coastal Geospatial Data Project (http://coastalgeospatial.noaa.gov).

Table E-1: TSS Concentrations (mg/L) Estimated by DCP, Southeastern Estuaries

Estuary	DCP	Base TSS (mg/l)
Albemarle/Pamlico Sound, NC, VA	0.14	33.7
Altamaha River, GA	0.37	82.8
Biscayne Bay, FLA	0.4	0.2
Bogue Sound, NC	1.47	3.0
Broad River, SC	4.92	16.3
Cape Fear River, NC	0.61	69.7
Charleston Harbor, SC	0.44	2.5
Indian River, FL	1.02	3.4
N and S Santee Rivers, SC	2.91	456.8
New River, NC	7.68	15.3
Ossabaw Sound, GA	1.99	81.0
Savannah River, SC, GA	0.43	61.2
St Catherines/Sapelo Sound, GA	7.56	17.2
St Helena Sound, SC	0.95	10.8
St Johns River, FL	0.83	13.3
St. Andrews/St. Simons Sound, GA	2.43	47.4
Winyah Bay, SC, NC	0.39	109.0

Table E-2: TSS Concentrations (mg/L) Estimated by DCP, Gulf of Mexico Estuaries

Estuary	DCP	Base TSS (mg/l)
Apalachee Bay, FL, GA	0.36	8.2
Apalachicola Bay, FL	0.17	27.7
Arkansas Bay, TX	6.02	152.7
Atchafalaya and Vermillion Bays, LA	0.04	254.8
Brazos River, TX	1.11	598.3
Calcasieu Lake, LA	1.18	48.3
Charlotte Harbor, FL	0.76	15.1
Choctawhatchee Bay, FL, AL	0.7	51.7
Corpus Cristi Inner Harbor, TX	4.67	555.0
Galveston Bay, TX	0.4	93.4

Table E-2: TSS Concentrations (mg/L) Estimated by DCP, Gulf of Mexico Estuaries

Estuary	DCP	Base TSS (mg/l)
Laguna Madre, TX	0.34	21.7
Matagorda Bay, TX	0.81	212.0
Mississippi River, LA, MS	0.01	198.4
Mississippi Sound, MS, LA, AL	0.17	127.5
Mobile Bay, AL	0.08	87.7
Pensacola Bay, FL	0.46	44.1
Perdido Bay, FL, AL	2.89	34.5
Sabine Lake, LA, TX	0.38	76.3
San Antonio Bay, TX	1.30	833.3
St Andrew Bay, FL	0.76	2.2
Suwanee River, FL	0.38	24.5
Tampa Bay, FL	1.03	10.4
Ten Thousand Islands, FL	1.94	1.6

Table E-3: TSS Concentrations (mg/L) Estimated by DCP, Northeastern Estuaries

Estuary	DCP	Base TSS (mg/l)
Barnegat Bay, NJ	1.36	3.7
Blue Hill Bay, ME	1.03	1.7
Buzzards Bay, MA	1.04	3.8
Cape Cod Bay, MA	0.69	0.5
Casco Bay, ME	0.61	3.1
Chesapeake Bay, VA, MD, DE, PA, DC	0.072	91.6
Chincoteague Bay, MD, VA	3.08	2.0
Delaware Bay, DE, NJ, PA, MD	0.14	40.0
Englishman Bay, ME	0.92	4.7
Gardiners Bay, NY	1.77	2.0
Great Bay, NE, NH	1.54	7.6
Great South Bay, NY	5.07	12.0
Hudson River/Raritan Bay, NY, NJ, MA, CT	0.2	52.4
Long Island Sound, NY, CT, MA	0.054	9.7
Massachusetts Bay, MA	0.27	2.3
Merrimack River, NH., MA	1.01	32.0
Muscongus Bay, ME	2.25	6.5
Narragansett Bay, MA, RI	0.52	8.2
Narragauges Bay, ME	1.54	4.2
Passaquoddy Bay, ME	0.27	1.5
Penobscot Bay, ME	0.13	7.1
Saco Bay, ME, NH	0.45	4.2
Sheepscop Bay, ME, NH	0.088	0.2

Table E-4: TSS Concentrations (mg/L) Estimated by DCP, West Coast Estuaries

Estuary	DCP	Base TSS (mg/l)
Alsea River, OR	2.086	37.4
Columbia River, WA, OR	0.032	77.0
Coos Bay, OR	2.27	50.9
Eel River, CA	0.39	65.3
Grays Harbor, WA	0.278	21.2
Humboldt Bay, CA	8	38.7
Klamath River, OR, CA	0.31	79.8
Monterey Bay, CA	1.05	148.2
Nehalam River, OR	1.556	38.5
Netarts Bay, OR	12.564	13.0
Puget Sound, WA	0.039	9.7
Rogue River, OR	0.556	125.4
San Diego Bay, CA	12.31	88.9
San Francisco Bay, CA	0.104	89.3
San Pedro Bay, CA	4.41	104.0
Santa Monica Bay, CA	1.371	30.0
Siletz Bay, OR	2.424	42.8
Siuslaw River, OR	1.853	46.5
Tillamook Bay, OR	1.049	21.3
Umpqua River, OR	0.793	110.6
Willapa Bay, WA	0.466	12.1
Yaquina Bay, OR	3.356	25.7

In the final analysis, the DCP approach was only used in 43 reaches without flow predictions. The number of reaches in each estuary where the DCP approach was used to calculate TSS concentrations is shown in *Table E-5* through *Table E-8* .

Table E-5: Total Number Reaches in the Analysis located in Southeastern Estuaries that Use DCP to Calculate TSS by Estuary

Estuary	DCP	Number of Reaches
Albemarle/Pamlico Sound, NC, VA	0.14	2
Altamaha River, GA	0.37	0
Bogue Sound, NC	1.47	0
Broad River, SC	4.92	0
Cape Fear River, NC	0.61	0
Charleston Harbor, SC	0.44	1
Indian River, FL	1.02	0
N and S Santee Rivers, SC	2.91	1
New River, NC	7.68	1
Ossabaw Sound, GA	1.99	1
Savannah River, SC, GA	0.43	0
St Catherines/Sapelo Sound, GA	7.56	0
St Helena Sound, SC	0.95	0
St Johns River, FL	0.83	2
Winyah Bay, SC, NC	0.39	1

Note: Only estuaries with reaches included in the analysis are listed in the table.

Table E-6: Total Number Reaches in the Analysis located in Gulf of Mexico Estuaries that Use DCP to Calculate TSS by Estuary

Estuary	DCP	Number of Reaches
Apalachicola Bay, FL	0.17	0
Charlotte Harbor, FL	0.76	0
Choctawhatchee Bay, FL, AL	0.7	2
Galveston Bay, TX	0.4	0
Pensacola Bay, FL	0.46	0
Perdido Bay, FL, AL	2.89	0
Sabine Lake, LA, TX	0.38	0
St Andrew Bay, FL	0.76	1
Suwanee River, FL	0.38	0
Tampa Bay, FL	1.03	2

Note: only estuaries with reaches included in the analysis are listed in the table.

Table E-7: Total Number Reaches in the Analysis located in Northeastern Estuaries that Use DCP to Calculate TSS by Estuary

Estuary	DCP	Number of Reaches
Barnegat Bay, NJ	1.36	1
Buzzards Bay, MA	1.04	0
Cape Cod Bay, MA	0.69	0
Casco Bay, ME	0.61	1
Delaware Bay, DE, NJ, PA, MD	0.14	0
Gardiners Bay, NY	1.77	0
Great Bay, NE, NH	1.54	0
Great South Bay, NY	5.07	0
Hudson River/Raritan Bay, NY, NJ, MA, CT	0.2	2
Long Island Sound, NY, CT, MA	0.054	1
Massachusetts Bay, MA	0.27	0
Narragansett Bay, MA, RI	0.52	2
Saco Bay, ME, NH	0.45	1
Sheepscop Bay, ME, NH	0.088	0

Note: Only estuaries with reaches included in the analysis are listed in the table.

Table E-8: Total Number Reaches in the Analysis located in West Coast Estuaries that Use DCP to Calculate TSS by Estuary

Estuary	DCP	Number of Reaches
Alsea River, OR	2.086	0
Coos Bay, OR	2.27	5
Grays Harbor, WA	0.278	3
Nehalam River, OR	1.556	0
Netarts Bay, OR	12.564	0
Puget Sound, WA	0.039	4
Rogue River, OR	0.556	1
San Diego Bay, CA	12.31	1
San Francisco Bay, CA	0.104	2
San Pedro Bay, CA	4.41	1
Santa Monica Bay, CA	1.371	0
Siletz Bay, OR	2.424	0
Siuslaw River, OR	1.853	0
Tillamook Bay, OR	1.049	2
Umpqua River, OR	0.793	0
Willapa Bay, WA	0.466	1
Yaquina Bay, OR	3.356	0

Note: Only estuaries with reaches included in the analysis are listed in the table.

Appendix F – TSS Subindex Curve Parameters

This appendix provides the ecoregion-specific parameters used in estimating the TSS water quality subindex, as follows:

If [TSS] ≤ TSS_{100}	Subindex = 100
If TSS_{100} < [TSS] ≤ TSS_{10}	Subindex = a exp(b [TSS])
If [TSS] > TSS_{10}	Subindex = 10

where [TSS] is the measured concentration and TSS_{10}, TSS_{100}, a, and b are specified in *Table F-1*.

Table F-1: TSS Subindex Curve Parameters, by Ecoregion

ID	Ecoregion Name	a	b	TSS_{100}	TSS_{10}
10.1.2	Columbia Plateau	126.56	-0.0038	63	668
10.1.3	Northern Basin and Range	112.42	-0.0007	160	3,457
10.1.4	Wyoming Basin	123.36	-0.001	220	2,513
10.1.5	Central Basin and Range	121.22	-0.0018	109	1,386
10.1.6	Colorado Plateaus	144.44	-0.001	363	2,670
10.1.7	Arizona/New Mexico Plateau	126.76	-0.0004	668	6,349
10.1.8	Snake River Plain	146.39	-0.0027	142	994
10.2.1	Mojave Basin and Range	119.34	-0.0015	121	1,653
10.2.2	Sonoran Desert	112.39	-0.0002	567	12,097
10.2.4	Chihuahuan Desert	214.39	-0.0005	1,419	6,130
11.1.1	California Coastal Sage, Chaparral, and Oak Woodlands	127.97	-0.0012	205	2,124
11.1.2	Central California Valley	171.86	-0.0044	122	646
11.1.3	Southern and Baja California Pine-Oak Mountains	115.12	-0.0007	197	3,491
12.1.1	Madrean Archipelago	261.35	-0.0005	2,053	6,527
13.1.1	Arizona/New Mexico Mountains	120.98	-0.0004	477	6,233
15.4.1	Southern Florida Coastal Plain	116.95	-0.0405	4	61
5.2.1	Northern Lakes and Forests	157.76	-0.0233	20	118
5.2.2	Northern Minnesota Wetlands	154.99	-0.0186	24	147
5.3.1	Northern Appalachian and Atlantic Maritime Highlands	174.99	-0.0261	21	110
5.3.3	North Central Appalachians	245.15	-0.0176	51	182
6.2.10	Middle Rockies	144.64	-0.0038	98	703
6.2.11	Klamath Mountains	238.9	-0.0068	129	467
6.2.12	Sierra Nevada	185.36	-0.0116	53	252
6.2.13	Wasatch and Uinta Mountains	124.28	-0.0014	160	1,800
6.2.14	Southern Rockies	153.42	-0.0031	140	881
6.2.15	Idaho Batholith	184.23	-0.0142	43	205
6.2.3	Columbia Mountains/Northern Rockies	180.7	-0.0168	35	172
6.2.4	Canadian Rockies	396.62	-0.0308	45	119
6.2.5	North Cascades	240.95	-0.0193	46	165
6.2.7	Cascades	192.94	-0.0181	36	164
6.2.8	Eastern Cascades Slopes and Foothills	178.82	-0.0145	40	199
6.2.9	Blue Mountains	148.35	-0.0037	107	729
7.1.7	Strait of Georgia/Puget Lowland	181.06	-0.0224	27	129
7.1.8	Coast Range	174.78	-0.0114	49	251

Table F-1: TSS Subindex Curve Parameters, by Ecoregion

ID	Ecoregion Name	a	b	TSS_{100}	TSS_{10}
7.1.9	Willamette Valley	210.3	-0.0114	65	267
8.1.1	Eastern Great Lakes and Hudson Lowlands	144.62	-0.0104	36	257
8.1.2	Lake Erie Lowland	112.79	-0.0049	25	494
8.1.3	Northern Appalachian Plateau and Uplands	322.68	-0.0113	103	307
8.1.4	North Central Hardwood Forests	148.68	-0.0108	37	250
8.1.5	Driftless Area	117.97	-0.0012	141	2,057
8.1.6	S. Michigan/N. Indiana Drift Plains	191.44	-0.0143	46	206
8.1.7	Northeastern Coastal Zone	158.48	-0.0164	28	168
8.1.8	Maine/New Brunswick Plains and Hills	156.02	-0.025	18	110
8.1.10	Erie Drift Plain	133.08	-0.0037	78	700
8.2.1	Southeastern Wisconsin Till Plains	121.34	-0.0042	46	594
8.2.2	Huron/Erie Lake Plains	145.17	-0.0058	65	461
8.2.3	Central Corn Belt Plains	187.95	-0.0033	191	889
8.2.4	Eastern Corn Belt Plains	235.18	-0.003	282	1,053
8.3.1	Northern Piedmont	175.82	-0.0042	135	683
8.3.2	Interior River Valleys and Hills	149.68	-0.0013	303	2,081
8.3.3	Interior Plateau	220.47	-0.0037	217	836
8.3.4	Piedmont	224.11	-0.0048	169	648
8.3.5	Southeastern Plains	205.3	-0.0085	85	356
8.3.6	Mississippi Valley Loess Plains	492.49	-0.0048	333	812
8.3.7	South Central Plains	184.36	-0.0045	136	648
8.3.8	East Central Texas Plains	162.32	-0.0013	362	2,144
8.4.1	Ridge and Valley	186.83	-0.0063	99	465
8.4.2	Central Appalachians	166.76	-0.0062	82	454
8.4.3	Western Allegheny Plateau	183.67	-0.0032	190	910
8.4.4	Blue Ridge	216.16	-0.0087	89	353
8.4.5	Ozark Highlands	175.16	-0.0018	317	1,591
8.4.6	Boston Mountains	329.77	-0.0062	193	564
8.4.7	Arkansas Valley	283.25	-0.004	261	836
8.4.8	Ouachita Mountains	212.77	-0.0048	157	637
8.4.9	Southwestern Appalachians	207.09	-0.0071	103	427
8.5.1	Middle Atlantic Coastal Plain	182.17	-0.0178	34	163
8.5.2	Mississippi Alluvial Plain	131.35	-0.0029	93	888
8.5.3	Southern Coastal Plain	138.62	-0.0144	23	183
8.5.4	Atlantic Coastal Pine Barrens	283.76	-0.0463	23	72
9.2.1	Aspen Parkland/Northern Glaciated Plains	136.43	-0.0005	640	5,226
9.2.2	Lake Manitoba and Lake Agassiz Plain	174.13	-0.0042	131	680
9.2.3	Western Corn Belt Plains	135.01	-0.0009	347	2,892
9.2.4	Central Irregular Plains	201.19	-0.001	673	3,002
9.3.1	Northwestern Glaciated Plains	133.98	-0.0006	483	4,325
9.3.3	Northwestern Great Plains	130.6	-0.0004	636	6,424
9.3.4	Nebraska Sand Hills	289.85	-0.0066	162	510
9.4.1	High Plains	125.61	-0.0005	507	5,061
9.4.2	Central Great Plains	156.84	-0.0005	925	5,505
9.4.3	Southwestern Tablelands	137.77	-0.0003	1,280	8,743
9.4.4	Flint Hills	270.93	-0.0009	1,084	3,666
9.4.5	Cross Timbers	134.97	-0.0006	523	4,337
9.4.6	Edwards Plateau	173.77	-0.001	544	2,855

Table F-1: TSS Subindex Curve Parameters, by Ecoregion

ID	Ecoregion Name	a	b	TSS_{100}	TSS_{10}
9.4.7	Texas Blackland Prairies	134.23	-0.0005	624	5,194
9.5.1	Western Gulf Coastal Plain	124.47	-0.0025	88	1,009
9.6.1	Southern Texas Plains/Interior Plains and Hills with Xerophytic Shrub and Oak Forest	166.67	-0.0003	1,602	9,378

Appendix G – Meta-Analysis Results

EPA used function-based benefit transfer to estimate benefits of surface water quality improvements due to reductions in sediment runoff from construction sites under the regulatory analysis options considered for the analysis of the regulation. The benefit function was derived using meta-analysis, following the general approach of Johnston et al. (2005), Shrestha et al. (2007), and others, following conceptual methods outlined by Bergstrom and Taylor (2006). The recent literature has given increasing emphasis to the potential use of meta-analysis to conduct and inform function-based benefit transfer (Johnston et al. 2005; Bergstrom and Taylor 2006; Rosenberger and Stanley 2006; Shrestha et al. 2007). For the present analysis, the meta-regression model was based on a model specification and data developed originally for the 316(b) Phase II Cooling Water Intake rule, and extended to generate a benefit function more suitable for assessing changes in ambient water quality from reducing sediment discharges. Chapter A12, “Methods for Estimating Non-use Benefits,” in the Regional Analysis document for the final Phase II rule provides details on the original meta-analysis (USEPA 2004d); revisions are detailed below. These revisions included adding new studies to the metadata, re-estimating the willingness-to-pay (WTP) function to better account for ecological services potentially affected by sediments and nutrients, and testing additional functional forms and statistical approaches.

As stated by Rosenberger and Johnston (2007, p. 1-2):

> One of the primary advantages of meta-analysis as a benefit transfer tool relates to its capacity to allow more appropriate adjustments of welfare measures based on patterns observed in the literature. Within a benefit transfer context, transfer error is often inversely related to the correspondence between a study site and a policy site among various dimensions (Rosenberger and Phipps 2007). The probability of finding a good fit between a single (or multiple) study site and a policy site, however, is usually low (Boyle and Bergstrom 1992; Spash and Vatn 2006). If, on the other hand, empirical studies contribute to a body of WTP estimates (i.e., metadata), and if empirical value estimates are systematically related to variations in resource, study, and site characteristics, then meta-regression analysis may provide a viable tool for estimating a more universal transfer function with distinct advantages over unit value or other function-based transfer methods (Johnston et al. 2003; Rosenberger and Loomis 2000a; Rosenberger and Stanley 2006). More specifically, Rosenberger and Phipps (2007) posit a meta-valuation function as the envelope of a set of empirically-defined valuation functions reported in the literature.

In the present case, EPA identified 45 valuation studies that use stated preference techniques to elicit benefit values for water quality improvements. To examine the relative influence of study, economic, and resource characteristics on WTP for improving surface water quality, the Agency conducted a regression-based meta-analysis of 115 estimates of WTP for water quality improvements, provided by the 45 original studies. Analytic methods and model specifications follow established methods in the published literature (e.g., Johnston et al. 2005; Bergstrom and Taylor 2006). The estimated econometric model is used as the basis of a function-based benefit transfer, to calculate WTP for improving water quality in waterbodies affected by sediment runoff from construction sites.

The following discussion summarizes the results of EPA’s meta-analysis of surface water valuation studies and the use of the resulting benefit function for transfer.

G.1 Literature Review of Water Resource Valuation Studies

As outlined in the introduction, EPA conducted a meta-analysis of water resource valuation studies to examine the relative influence of study, economic, and resource characteristics on total WTP for water quality improvements. The Agency analyzed 45 studies, published between 1981 and 2008, that applied generally accepted, standard valuation methods to determine total (including use and nonuse) values associated with aquatic habitat improvements. These 45 studies all used stated preference techniques to assess WTP, but varied in other respects, including the survey administration methods used, the specific environmental change valued, and the geographic region affected by the environmental changes. Studies using stated preference approaches are preferred to studies using revealed preference approaches, because they elicit total household WTP (including use and nonuse values). Revealed preference studies allow to estimate use values only. Data from the 45 studies result in a total of 115 observations for the meta-analysis because 30 studies provide more than one usable estimate of total WTP for aquatic habitat improvements.

When constructing metadata for subsequent meta-analysis, analysts must determine the optimal scope of the metadata (Rosenberger and Johnston 2007), interpreted as the exact definition of the dependent variable in the meta-regression model, which, in turn, defines the set of source studies that can be considered for inclusion in the metadata. The primary tradeoff is often between maintaining close similarity among dependent variables versus including additional information (i.e., observations) in the metadata. Similarity in dependent variable definition and study attributes within the metadata can be important for two reasons. First, theory may dictate that certain types of estimated values are not strictly comparable (e.g., Hicksian compensating variation from a stated preference model versus Marshallian consumer surplus from a travel cost model). Second, model fit may be improved by narrowing the metadata, for example to include only valuation studies that use a particular valuation approach (e.g., stated preference methods, as in Johnston et al. (2005); or travel cost model estimates, as in Smith and Kaoru (1990)). Such study selection issues may be framed in terms of a requirement that at a minimum, studies included in metadata satisfy both *commodity consistency* and *welfare consistency* (Bergstrom and Taylor 2006). The former implies that "the commodity (Q) being valued should be approximately the same within and across studies" (Bergstrom and Taylor 2006, p. 353). The latter implies that "measures of WTP within and across studies … should represent the same … welfare change measure, or ex-post calibrations [are] made to account for theoretical differences between welfare change measures" (Bergstrom and Taylor 2006, p. 355).

The requirement of welfare consistency implies that—outside of preference calibration approaches that explicitly account for theoretical differences between welfare constructs (e.g., Smith et al. 2002)—meta-analyses should not combine data representing theoretically distinct welfare measures. For example, the ad hoc combination of stated and revealed data for meta-analysis would generally violate the condition of welfare consistency. Although one may introduce independent variables in regression models (typically dummy variable intercept shifters) to account for such theoretical differences, Smith and Pattanayak (2002) argue that such methods are unlikely to represent appropriate adjustments for fundamental differences in theoretical welfare measures.

G.1.1 Identifying Water Resource Valuation Studies

EPA identified surface water valuation studies used in the total WTP meta-analysis by conducting an in-depth search of the economic literature. EPA used a variety of sources and search methods to identify relevant studies:

- Review of EPA's research and bibliographies dealing with non-market benefits associated with water quality changes

- Systematic review of recent issues of key resource economics journals (e.g., Land Economics, Marine Resource Economics, Journal of Environmental Economics and Management)
- Searches of online reference and abstract databases (e.g., Environmental Valuation Resource Inventory (EVRI), Benefits Use Valuation Database (BUVD), AgEcon Search)
- Visits to home pages of authors known to have published stated preference studies and/or water quality valuation research
- Searches of Web sites of agricultural and resource economics departments at several colleges and universities
- Searches of Web sites of organizations and agencies known to publish environmental and resource economics valuation research (e.g., Resources for the Future (RFF), National Center for Environmental Economics (NCEE)).

From this review, EPA identified approximately 300 surface water valuation studies that were potentially relevant for this analysis and compiled a bibliographic database to organize the literature review process. Sixty-seven of these studies met the criteria identified for inclusion in the meta-analysis.[23] These criteria were designed to ensure both commodity and welfare consistency as noted above, and include:

- **Specific amenity valued:** Selected studies were limited to those in which the environmental quality change being valued affects ecological services provided by surface waterbodies, including aquatic life support, recreational activities (such as fishing, boating, and swimming), and nonuse value
- **Values estimated:** Selected studies were limited to those that used stated preference techniques to elicit household WTP.
- **Study location:** Selected studies were limited to those that surveyed U.S. and Canadian populations to value resources
- **Research methods:** Selected studies were limited to those that applied research methods supported by journal literature.

The Agency compiled extensive information from the 67 selected studies. Of these studies, 45 were utilized in the model estimation. Reasons for the difference between the total number of studies in the final metadata (67) and studies represented in the final model (45) include unavailability of information for certain key regressors for all studies.

The tradeoff between the number of regressors or independent variables that may be included in a meta-regression analysis (K) and the number of studies that are included in the metadata (N) is a fundamental tradeoff in most meta-analyses in the valuation literature (Moeltner et al. 2007). That is, if a study considered for inclusion in the metadata does not provide information for a certain regressor that analysts might wish to include in the meta-regression, analysts must generally choose between omitting the regressor from the meta-regression or omitting the study from the metadata. As a result, researchers wishing to increase the number of explanatory variables in meta-regression models (increasing K) often do so at the cost of reductions in the number of studies or observations in the metadata (reducing N).

[23] The remaining studies were either earlier unpublished versions or slightly modified versions of the included studies, focused on water resources outside of the United States and Canada, used secondary research methods, or valued environmental quality changes that were not directly linked to changes in water quality (e.g., change in recreational catch rates).

Conversely, increases in the number of observations in meta-regression models are sometimes only possible if one reduces the number of independent variables in the model. Hence, there is a tradeoff between the quantity of information in the metadata (i.e., the number of observations or studies) and the possible risk of omitted variables bias due to the omission of influential regressors. See Moeltner et al. (2007) for additional discussion of this *"N* versus *K"* tradeoff in meta-analysis.

The complete data set used in the meta-analysis is provided in the docket for the regulation (DCN 6-7900), and includes the following information:

- Full study citation
- Study location
- Sample data and description (e.g., size, response rate, income)
- Resource characteristics (e.g., affected waterbody type, recreational uses, baseline quality)
- Description of environmental quality change, including geographic scale, affected species, and affected recreational uses (e.g., water quality change from fishable to boatable)
- Quantitative measure of environmental quality change measured in terms of improvements in Water Quality Index (WQI)[24] and/or percentage reduction in pollutant concentration
- Study WTP values updated to 2007 dollars
- WTP estimation characteristics (i.e., parametric versus non-parametric, inclusion of protest bids and outlier bids, WTP description).

G.1.2 Description of Studies Selected for Total WTP Meta-Analysis

The 45 studies that EPA used in the total WTP meta-analysis were conducted between 1981 and 2008, and applied standard, generally accepted stated preference valuation methods to assess WTP. Studies were excluded if they did not conform to general tenets of economic theory, or if they applied methods not generally accepted in the literature.

All selected studies focus on environmental quality changes that affect surface water resources in the United States. Beyond this general similarity, the studies vary in several respects. Differences include the specific environmental change valued, scale of environmental improvement, geographic region affected by environmental changes, types of values estimated, survey administration methods, demographics of the survey sample, and statistical methods employed. The 45 studies include 25 journal articles, 6 reports, 5 Ph.D. dissertations, 7 academic or staff papers, 1 book, and 1 master's thesis.

The 45 studies selected for the meta-analysis provided 115 observations in the final data set because multiple estimates of WTP were available from 30 studies. The availability of multiple observations from single studies is common in meta-analyses of this type (e.g., Bateman and Jones 2003; Johnston et al. 2005). Some of the characteristics that allowed multiple observations to be derived from a single study include the extent of the amenity change, the respondent population type, elicitation method(s),

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[24] Additional details on the WQI and the use of the WQI in survey instruments are provided by McClelland (1974), Vaughan (1986), and Mitchell and Carson (1989, p. 342). This index is linked to specific pollutant levels, which in turn are linked to presence of aquatic species and suitability for particular recreational uses. The WQI allows the use of objective water quality parameters (e.g., dissolved oxygen concentrations) to characterize ecosystem services or uses provided by a given waterbody.

waterbody type, number of waterbodies affected, recreational activities affected by the quality change, and species affected by the quality change. These variations are often due to experimental design driven by the key research questions or hypotheses. *Table G-1* lists key study and resource characteristics and indicates the number of observations derived from each study.

Surveys in 26 studies were administered by mail; 9 studies collected information through personal interviews in the home, onsite, or in a centralized location, 9 surveys were conducted by telephone, and 1 conducted by computer administration. Study sample sizes range from 96 to 4,033 responses.

Table G-1: Selected Summary Information for Studies

Author(s) and Year	Observations	State	Waterbody Type	Type of Water Quality Improvement	Affected Recreational Uses
Aiken (1985)	1	CO	all freshwater	general water quality	fishing
Anderson and Edwards (1986)	1	RI	salt pond/marshes	general water quality	fishing and swimming
Azevedo et al. (2001)	5	IA	lake	nutrients	fishing and swimming
Bockstael et al. (1988)	1	DC, MD, VA	estuary	general water quality; phosphorus and nitrogen	swimming, beach, boating, fishing, outings
Bockstael et al. (1989)	2	MD	estuary	general water quality	swimming
Breffle et al. (1999)	2	WI	estuary	general water quality	fishing
Cameron (1988)	1	TX	Estuary	general water quality	fishing
Cameron and Huppert (1989)	1	CA	river/stream	wildlife habitat	game fishing
Carson and Mitchell (1993)	4	National	multiple	general water quality	boating; fishing; swimming
Carson et al. (1994)	2	CA	estuary	DDT and PCBs	fishing
Clonts and Malone (1990)	3	AL	river/stream	general water quality	multiple uses
Croke et al. (1986-87)	9	IL	river/stream	general water quality	multiple uses
De Zoysa (1995)	2	OH	river and lake	sediment and nutrients; wildlife habitat	multiple uses
Desvousges et al. (1987)	12	PA	river/stream	general water quality	multiple uses
Hayes et al. (1992)	2	RI	estuary	general water quality	swimming; fishing
Herriges and Shogren (1996)	2	IA	lake	general water quality	boating and fishing
Hite (2002)	2	MS	river/stream	general water quality	multiple uses
Huang et al. (1997)	2	NC	estuary	general water quality	fishing
Hushak and Bielen (1999)	2	OH, MI	river/stream	general water quality	multiple uses
Kaoru (1993)	1	MA	salt pond/marshes	fecal coliform	fishing
Lant and Roberts (1990)	3	IA, IL	river/stream	sediment	boating, fishing, and swimming
Lant and Tobin (1989)	9	IA, IL	river/stream	general water quality	boating, fishing
Lichtkoppler and Blaine (1999)	1	OH	multiple	PCBs and general water quality	all recreational uses
Lindsey (1994)	8	MD	estuary	nutrients	multiple uses
Lipton (2004)	1	MD	estuary	general water quality	boating
Loomis (1996)	1	WA	river/stream	general water quality	fishing
Loomis et al. (2000)	2	CO	river/stream	general water quality	multiple uses
Lyke (1993)	2	WI	lake	general water quality	fishing

Table G-1: Selected Summary Information for Studies

Author(s) and Year	Observations	State	Waterbody Type	Type of Water Quality Improvement	Affected Recreational Uses
Matthews et al. (1999)	2	MN	river/stream	phosphorus	boating and fishing
Olsen et al. (1991)	3	ID, MT, OR, WA	river/stream	wildlife habitat	fishing
Opaluch et al. (1998)	1	NY	estuary	general water quality	shellfishing
Roberts and Leitch (1997)	1	MN, SD	lake	general water quality	multiple uses
Rowe et al. (1985)	1	CO	river/stream	general water quality	boating, fishing, and swimming
Sanders et al. (1990)	4	CO	river/stream	general water quality	swimming
Schulze et al. (1995)	2	MT	river and lake	hazardous pollutants	boating, fishing, and swimming
Shrestha and Alavalapati (2004)	2	FL	multiple	phosphorus and wildlife habitat	multiple uses
Stumborg et al. (2001)	2	WI	lake	phosphorus	multiple uses
Sutherland and Walsh (1985)	1	MT	river and lake	general water quality	swimming
Viscusi et al. (2008)	2	National	river and lake	general water quality	multiple uses
Welle (1986)	6	MN	all freshwater	acid rain	game fishing and wildlife viewing
Wey (1990)	2	RI	salt pond/ marshes	general water quality	Other
Whitehead et al. (2002)	1	NC	river/stream	general water quality	fishing, boating, swimming
Whitehead and Groothuis (1992)	3	NC	river/stream	sediment and nutrients	multiple uses
Whitehead et al. (1995)	2	NC	estuary	general water quality	boating, fishing, and swimming
Whittington et al. (1994)	1	TX	estuary	heavy metals and pesticides	multiple uses

The Agency's review of the relevant economic literature showed that available surface water valuation studies focus primarily on general water quality rather than specific pollutants or changes. Even in cases in which specific pollutants are the primary policy issue, the stated preference surveys from which welfare estimates are derived often characterize water quality changes only in general (i.e., non-pollutant specific) terms. Hence, the associated welfare measures are conditioned on this general description. Of the 45 studies, 26 presented only WTP values for changes in general water quality (approximately 60 percent). Of the studies that did address specific changes, eight specified nutrients and/or sediment, four addressed hazardous pollutants including heavy metals and pesticides, three addressed wildlife habitat, one addressed acid deposition, one addressed fecal coliform bacteria, one presented values for both changes in general water quality and nutrient reductions, and one presented values for both changes in general water quality and wildlife habitat. Preliminary model estimates showed no evidence that the type of pollutant considered had a statistically significant influence on WTP across and within studies. For this reason, EPA used a standardized scale to define both the baseline water quality and the water quality change valued in the original study. Additional details are provided below.

From these 45 studies, the Agency compiled a data set for the meta-analysis of total WTP values. EPA specified a regression model based on these data to estimate a range of total household benefits for surface water and aquatic habitat improvements. General empirical methods follow those outlined by Johnston et al. (2005), following standard approaches in the meta-analysis literature. The model and results are described in the next section.

G.2 Total WTP Meta-Analysis Regression Model and Results

EPA estimated both trans-log and semi-log meta-regression models based on 115 WTP estimates for improvements in water resources, derived from 45 original studies.[25] These metadata, the model specification, model results, and interpretation of the results are described in the following sections. EPA, however, notes that only the trans-log model is used in the analysis of benefits from reduced sediment discharges. The alternative specification (semi-log) is presented for comparative purposes only.

In a frequently cited work, Glass (1976) characterizes meta-analysis as "the statistical analysis of a large collection of results for individual studies for the purposes of integrating the findings. It provides a rigorous alternative to the casual, narrative discussion of research studies which is commonly used to make some sense of the rapidly expanding research literature" (p. 3; cited in Poe et al. 2001, p. 138). Meta-analysis is being increasingly explored as a potential means to estimate resource values in cases where original targeted research is impractical, or as a means to reveal systematic components of WTP (e.g., Smith and Osborne 1996; Santos 1998; Rosenberger and Loomis 2000a; Poe et al. 2001; Woodward and Wui 2001; Bateman and Jones 2003). While the literature often urges caution in the use and interpretation of benefit transfers for direct policy application (e.g., Desvousges et al. 1998; Poe et al. 2001; Navrud and Ready 2007), such methods are "widely used in the United States by government agencies to facilitate benefit-cost analysis of public policies and projects affecting natural resources" (Bergstrom and De Civita 1999). Transfers based on meta-analysis are likewise common in both the United States and Canada (Bergstrom and De Civita 1999; Bergstrom and Taylor 2006).

Depending on the suitability of available data, a meta-analysis can provide a superior alternative to the calculation and use of a simple arithmetic mean WTP over the available observations, as it allows estimation of the systematic influence of study, economic, and natural resource attributes on WTP (USEPA 2000b; Rosenberger and Phipps 2007; Shrestha et al. 2007). The primary advantage of a regression-based (statistical) approach is that it accounts for differences among study characteristics that may contribute to changes in WTP, to the extent permitted by available data (Johnston et al. 2005; Rosenberger and Phipps 2007). An additional advantage is that meta-analysis can reveal systematic factors influencing WTP, allowing analysts to assess whether, for example, WTP estimates are (on average) sensitive to scope (Smith and Osborne 1996).

There is, however, some controversy regarding whether regression-based meta-analyses should be used for direct benefit transfer. Many contemporary sources in the literature note the potential ability of regression-based meta-analyses to generate benefit functions better able to adjust and forecast benefits at policy sites in question, and either explicitly or implicitly favor the use of meta-analysis over alternative benefit transfer approaches (e.g., Johnston et al. 2005; Bergstrom and Taylor 2006; Moeltner et al. 2007; Rosenberger and Johnston 2007; Rosenberger and Phipps 2007; Shrestha et al. 2007). EPA (USEPA 2000b) characterizes meta-analysis as "the most rigorous" benefit transfer method. In contrast, the EPA Science Advisory (2007) Board's "Advisory on EPA's Issues in Valuing Mortality Risk Reduction" recommends against the use of regression-based meta-analysis for VSL (value of statistical life) transfers, and other authors advise caution in such uses (Navrud and Ready 2007). The primary disagreement is whether it is appropriate to use meta-analysis results as a reduced form model to estimate benefits, and

[25] In its analysis of nonuse benefits for the final 316(b) Phase II rule, EPA also specified trans- and semi-log regression models similar to the models estimated for Phase II discussed in this section. See Chapter A12, "Methods for Estimating Non-use Benefits," in the Regional Analysis document for further details regarding both the log-log and semi-log regression models estimated in EPA's analysis of nonuse benefits for the final Phase II rule (USEPA 2004d); http://www.epa.gov/ost/316b/casestudy/final.htm).

whether the empirical ability of meta-analysis in many cases to generate benefit transfers with reduced transfer errors offsets the lack of an underlying theoretical model to "calibrate" benefit estimates across studies (cf. Smith and Pattanayak 2002; Smith et al. 2002). While the Agency recognizes this ongoing controversy, it notes that there are a large number of practitioners and publications supporting the use of regression-based meta-analysis for benefit transfer. It also removes the element of subjective judgment associated with selecting a single study or value for benefit transfer. Hence, the following model is presented as a means to provide a benefit function that capitalizes on the substantial information available for existing water quality valuation studies, notwithstanding potential concerns voiced by some regarding the use of meta-analyses for such purposes.

G.2.1 Metadata Total WTP Regression Model

Meta-analysis is largely an empirical, data-driven process, but one in which variable and model selection is guided by theory (Bergstrom and Taylor 2006). Given a reliance on information available from the underlying studies that comprise the metadata, meta-analysis models most often represent a middle ground between model specifications that would be most theoretically appropriate and those specifications that are possible given available data. Smith and Osborne (1996), Rosenberger and Loomis (2000a), Poe et al. (2001), Bateman and Jones (2003), Dalhuisen et al. (2003), Johnston et al. (2005, 2006), Bergstrom and Taylor (2006), Moeltner et al. (2007), and others provide insight into the mechanics of specifying and estimating meta-equations in resource economics applications.

Past meta-analyses have incorporated a range of different statistical methods, with none universally accepted as superior (Johnston et al. 2005). EPA followed recent work of Bateman and Jones (2003) and Johnston et al. (2005) in applying a multilevel model specification to the metadata, to address potential correlation among observations gathered from single studies. Also following prior work (e.g., Smith and Osborne 1996; Poe et al. 2001), EPA applied the Huber-White robust variance estimation. As described by Smith and Osborne (1996, p. 293), "this approach treats each study as the equivalent of a sample cluster with the potential for heteroskedasticity…across clusters." Weighted models are avoided following the arguments of Bateman and Jones (2003).[26]

To guide development of the model and variable specifications, EPA relied upon a set of general principles. These principles are designed to help prevent excessive data manipulations and other factors that may lead to misleading model results. The general principles include, all else being equal:

- Fewer and simpler data transformations are preferred to more extensive ones.
- In the absence of overriding theoretical considerations, continuous variables are generally preferred to discrete variables derived from underlying continuous distributions.
- Models should attempt to capture elements of the scope and scale of resource changes.
- Models should distinguish WTP associated with different types of resources and resource uses, particularly where relevant to the policy question at hand.
- Following the "weak structural utility theoretic" (WSUT) approach of Bergstrom and Taylor (2006, p. 352), exogenous structural constraints are avoided to afford the flexibility necessary

[26] For comparison, models were also estimated using ordinary least squares (OLS) with robust variance estimation, weighted least squares with robust variance estimation, and multilevel models with standard (non-robust) variance estimation. None of these models outperformed the illustrated model in terms of overall model significance and fit, and statistical significance of individual coefficients (see *Section G.2.2* for further details concerning the specification of the model).

to appropriately model empirical patterns that may not necessarily flow from an underlying theoretical modeling structure. The dependent variable in the meta-analysis is the natural logarithm of estimated household WTP for water quality improvements in aquatic habitat, as reported in each original study. For this analysis, original study values were adjusted to 2007$ based on the relative change in Consumer Price Index (CPI) from the study year to 2007. Total WTP over the sample ranged from $5.33 to $502.70, with a mean value of $83.09.

As noted above, two model specifications are estimated (cf. Johnston et al. 2005). For the first specification, all right-hand-side variables are linear, resulting in a standard semi-log functional form. The second specification is identical to the semi-log model, except for the specification of the explanatory variable measuring water quality change and baseline as natural logs. This results in a trans-log functional form, also common in empirical applications (Johnston et al. 2001, 2005). Both the semi-log and trans-log models have advantages related to (1) their fit to the data, (2) the intuitive results that are provided, and (3) their common use in the empirical valuation and meta-analysis literature (e.g., Smith and Osborne 1996; Santos 1998; Johnston et al. 2001, 2005, 2006). The trans-log model, however, has the additional structural advantage that estimated WTP is necessarily zero when water quality change is also zero (cf. Johnston et al. 2001)—a property suggested by theory that analysts may wish to weight against model fit considerations when choosing a model for benefit estimation. While linear forms are also common in this literature (Rosenberger and Loomis 2000a, 2000b; Poe et al. 2001; Bateman and Jones 2003), specifications requiring more intensive data transformations (e.g., Box-Cox, log-log) are less common.

As noted in the preceding section, the metadata include independent variables characterizing specific details of the resource(s) valued such as the baseline resource conditions; the extent of resource improvements and whether they occur in estuarine or freshwater; the geographic region and scale of resource improvements (e.g., the number of waterbodies); resource characteristics (e.g., baseline conditions, the extent of water quality change, and ecological services affected by resource improvements); characteristics of surveyed populations (e.g., users, nonusers); and other specific details of each study. For ease of exposition, these variables are categorized into those characterizing (1) study and methodology, (2) surveyed populations, (3) geographic region and scale, and (4) resource improvements. Attributes included within each category are summarized below.

Study and methodology variables characterize such features as:

- The year in which a study was conducted
- The payment vehicle and elicitation format (e.g., discrete choice versus open-ended, voluntary versus non-voluntary, interview versus mail versus phone)
- WTP estimation methods and conventions (e.g., approaches to protest and outlier bids, use of parametric versus non-parametric statistical methods, estimation of mean or median WTP, the use of annual or lump-sum payments)
- Whether the original survey represented water quality changes using the WQI.

Surveyed populations variables characterize such features as:

- The average income of respondents
- Whether the survey specifically targeted nonusers.

Geographic region and scale variables characterize such features as:

- The number of waterbodies affected by the policy
- Whether the study considered water quality improvements in all waterbodies in a region
- The geographic area of the country in which the study was conducted.

Resource improvement variables characterize such features as:

- The extent of water quality change estimated as a difference between the baseline and post-change water quality index
- Baseline water quality index
- Those studies for which recreational uses such as fishing are specifically noted in the survey
- Aquatic species affected by resource improvements (e.g., game fish and shellfish)
- Those studies identifying large increases in fish populations (i.e., greater than 50 percent).

Although the interpretation and calculation of most independent variables requires little explanation, a few variables require additional detail. These include the variables characterizing surface water quality and its measurement. Many (23) observations in the metadata characterize quality changes using variants of the WQI (e.g., Mitchell and Carson 1989). This scale is linked to specific pollutant levels, which, in turn, are linked to the presence of aquatic species and recreational uses. However, some observations provide water quality measures using other, primarily descriptive, means that differ from the WQI.

To allow consistent comparisons of water quality change using a single scale, EPA mapped all water quality measures to the original WQI developed by McClelland (1974). WQI values were therefore developed for those studies that did not originally use this index. This scale was chosen for two reasons:

- WQI values are linked to specific pollutant levels including sediment concentrations, which, in turn, are linked to the presence of aquatic species and suitability for particular recreational uses. Therefore, the WQI can be used to link water quality changes from reduced sediment runoff to effects on human uses and support for aquatic species habitat. *Chapter 10* of this report provides detail on application of the WQI to estimating resource improvements from reduced sediment runoff from construction sites.
- A large number of the original studies in the metadata included WQI measures as "native" components of the original surveys. Hence, for these studies, no additional transformations were required.

While not all studies in the metadata included the WQI as a native survey component, in most cases the descriptions of water quality (present in the studies that did not apply the WQI) rendered mapping of water quality measures to the WQI straightforward. In cases where baseline and improved (or declined) water quality was not defined by suitability for recreational activities (e.g., boating, fishing, and swimming) or corresponding qualitative measures (e.g., poor, fair, good), EPA used descriptive information available from studies (e.g., amount/indication of the presence of specific pollutants, historical decline of the quality of the resource) to approximate the baseline level of water quality and the magnitude of the change.[27] For studies that valued discrete changes in the size of species populations, EPA characterized the baseline quality based on the current presence and prevalence of the species at

[27] For example, a study by Huang et al. (1997) described current water quality as degraded from 1981 levels in terms of reduced fish catches (60 percent) and reduced number of open shellfish beds (25 percent). However, because the water resource was still supporting recreational fishery, the baseline water quality was set to "fishable" on the WQI.

hand, and assumed population increases to correspond to modest increases in water quality in order to be conservative.[28] To account for the uncertainty involved in mapping those studies that are not based on the WQI, EPA introduced the binary variable *WQI*, which indicates those studies in which WQI measurements were an original component of the survey instrument. This approach is based largely on the published methods of Johnston et al. (2005), drawn from prior Agency work for the 316(b) Phase II Cooling Water Intake rule.

Variables incorporated in the final model are listed and described in *Table G-2*.

Table G-2: Variables and Descriptive Statistics for the Total WTP Regression Model

Variable	Description	Units and Measurement	Mean (Std. Dev.)
ln_WTP	Natural log of WTP for specified resource improvements.	Natural log of dollars (Range: 2.12 to6.22)	4.42 (0.75)
year_indx	Year in which the study was conducted, converted to an index by subtracting 1980.	Year index (Range: 1 to 28)	9.68 (6.42)
discrete	Binary (dummy) variable indicating that WTP was estimated using a discrete choice survey instrument.	Binary (Range: 0 or 1)	0.28 (0.45)
volunt	Binary (dummy) variable indicating that WTP was estimated using a payment vehicle described as voluntary as opposed to, for example, property taxes.	Binary (Range: 0 or 1)	0.05 (0.22)
mail	Binary (dummy) variable indicating that the survey was conducted through mail (default value for this dummy is a phone survey).	Binary (Range: 0 or 1)	0.48 (0.52)
lump_sum	Binary (dummy) variable indicating that payments were to occur on something other than an annual basis over a long period of time, such as property taxes. For example, some studies specified that payments would occur over a five-year period.	Binary (Range: 0 or 1)	0.13 (0.34)
nonparam	Binary (dummy) variable indicating that WTP was estimated using non-parametric methods.	Binary (Range: 0 or 1)	0.50 (0.50)
quality_ch	The change in mean water quality, specified on the WQI (McClelland 1974; Mitchell and Carson 1989). Defined as the difference between baseline and post-compliance quality. Where the original study (survey) did not use the WQI, EPA mapped water quality descriptions to analogous levels on the WQI to derive water quality change (see text).	WQI units (Range: 2.5 to 65)	21.3 (10.4)
lnquality_ch[1]	Natural log of the change in mean water quality (*quality_ch*), specified on the WQI (McClelland 1974; Mitchell and Carson 1989).	WQI units (Range: 0.92 to 4.17)	2.91 (0.59)
WQI	Binary (dummy) variable indicating that the original survey reported resource changes using a standard WQI.	Binary (Range: 0 or 1)	0.34 (0.48)
non-reviewed	Binary (dummy) variable indicating that the study was not published in a peer-reviewed journal.	Binary (Range: 0 or 1)	0.32 (0.47)
outlier_bids	Binary (dummy) variable indicating that outlier bids were excluded when estimating WTP.	Binary (Range: 0 or 1)	0.94 (0.24)
median_WTP	Binary (dummy) variable indicating that the study reported median, not mean, WTP.	Binary (Range: 0 or 1)	0.04 (0.20)
income	Mean income of survey respondents, either as reported by the original survey or calculated by EPA based on U.S. Census Bureau averages for the original surveyed region.	Dollars (Range: 34,955 to 158,347)	5,7049.59 (13,946.64)

28 For example, a study by Lyke (1993) describes the baseline conditions as follows: (1) “there are no naturally reproducing lake trout in Lake Michigan; all lake trout found there are from hatcheries.” (2) “Lake Superior stocks of self-reproducing lake trout were much reduced, but not wiped out, and both natural and hatchery-raised lake trout are found there.” These baseline conditions correspond to the “game-fishable” level on the WQI. The study estimates WTP for restoring natural populations of lake trout to the Wisconsin Great Lakes. Therefore, the expected change that will occur within the “game-fishable” category is likely to be small.

Table G-2: Variables and Descriptive Statistics for the Total WTP Regression Model

Variable	Description	Units and Measurement	Mean (Std. Dev.)
nonusers	Binary (dummy) variable indicating that the survey was implemented over a population of nonusers (default category for this dummy is a survey of any population that includes users).	Binary (Range: 0 or 1)	0.09 (0.29)
single_river[2]	Binary (dummy) variable indicating that resource change explicitly took place over a single river (default is a change in an estuary or that takes place on a national scale).	Binary (Range: 0 or 1)	0.23 (0.42)
single_lake[3]	Binary (dummy) variable indicating that resource change explicitly took place over a single lake (default is a change in an estuary or that takes place on a national scale).	Binary (Range: 0 or 1)	0.08 (0.28)
regional_fresh	Binary (dummy) variable indicating that resource change explicitly took place over an entire region such as a state (default is a change in an estuary or a change that takes place on a national scale).	Binary (Range: 0 or 1)	0.36 (0.48)
multiple_river	Binary (dummy) variable indicating that resource change explicitly took place over multiple rivers (default is a change in an estuary or that takes place on a national scale).	Binary (Range: 0 or 1)	0.06 (0.24)
salt_pond	Binary (dummy) variable indicating that resource change explicitly took place over multiple salt ponds (default is a change in an estuary or that takes place on a national scale).	Binary (Range: 0 or 1)	0.03 (0.18)
num_riv_pond	Number of rivers or salt ponds affected by policy; if unspecified *num_riv_pond* = 0. (In the present data, only studies addressing rivers and lakes specified >1 number of waterbodies. All others specified either 1 waterbody, or the number was unspecified.)	Number of specified rivers or ponds (Range: 0 to 15)	1.01 (2.98)
mr	Binary (dummy) variable indicating that the survey included respondents from more than one of the EPA regions.	Binary (Range: 0 or 1)	0.08 (0.26)
mp	Binary (dummy) variable indicating that the survey included respondents from the Mountain Plain region.[4]	Binary (Range: 0 or 1)	0.02 (0.13)
allmult	Binary (dummy) variable indicating that either all or multiple aquatic species are affected by the resource change.	Binary (Range: 0 or 1)	0.18 (0.39)
nonspec	Binary (dummy) variable indicating that the study did not specify what species would be affected by water quality improvements.	Binary (Range: 0 or 1)	0.42 (0.50)
fish_use	Binary (dummy) variable identifying studies in which changes in fishing use are specifically noted in the survey.	Binary (Range: 0 or 1)	0.56 (0.50)
fishplus	Binary (dummy) variable identifying studies in which a fish population or harvest change of 50 percent or greater is reported in the survey.	Binary (Range: 0 or 1)	0.08 (0.28)
baseline	Baseline water quality, specified on the WQI.	WQI units (Range: 1.61 to 5.2)	39.79 (20.37)
lnbase	Natural log of baseline water quality, specified on the WQI.	WQI units (Range: 5 to 70)	3.46 (0.80)

[1] The variable quality_ch is defined earlier in this table as the difference between baseline and post-compliance quality, specified on the WQI (Mitchell and Carson 1989).
[2] Examples of rivers and streams considered in the studies include the Columbia, Potomac, Elwha, Eagle, and Tar-Pamlico rivers.
[3] Includes one study that focused on a segment of the Lake Erie shoreline.
[4] The Mountain Plain region includes the following states: Colorado, Iowa, Kansas, Missouri, Montana, Nebraska, North Dakota, South Dakota, Utah, Wyoming.

G.2.2 Total WTP Regression Model and Results

As noted above, EPA estimated the meta-analysis regression using a multilevel, random-effects specification. This model follows the general approaches of Bateman and Jones (2003) and Johnston et al. (2005), among others. Multilevel (or hierarchical) models may be estimated as either random-effects or random-coefficients models, and are described in detail elsewhere (Singer 1998). The fundamental distinction between these and classical linear models is the two-part modeling of the equation error to account for hierarchical data. Here, the metadata are comprised of multiple observations per study, and there is a corresponding possibility of correlated errors among observations that share a common study or author.

The common approach to modeling such potential correlation is to divide the residual variance of estimates into two parts, a random error that is independently and identically distributed across all studies and for each observation, and a random effect that represents systematic variation related to each study. The model is estimated as a two-level hierarchy, with level one corresponding to WTP estimates (individual observations), and level two corresponding to individual studies. The random effect may be interpreted as a deviation from the mean equation intercept associated with individual studies (Bateman and Jones 2003). The model is estimated using a maximum likelihood estimator, assuming that random effects show a multivariate normal distribution. Following Bateman and Jones (2003), observations are unweighted. Covariances are obtained using the Huber-White covariance estimator (Smith and Osborne 1996). Random-effects models such as the multilevel model applied here are becoming increasingly standard in resource economics applications, and are estimable using a variety of readily available software packages.

A Note on Model Specification

As noted above, EPA considered two functional forms in this analysis: semi-log and trans-log. In both cases, the dependent variable is the natural log of estimated household WTP for surface water quality improvement, as shown in *Table G-2*. For Model One, all right-hand-side variables are linear, resulting in a semi-log functional form common in meta-analysis (e.g., Smith and Osborne 1996; Johnston et al. 2003). While linear forms are also common (Rosenberger and Loomis 2000a, 2000b; Poe et al. 2001; Bateman and Jones 2003), the semi-log and trans-log forms were chosen based on statistical performance and ability to capture curvature in the valuation function, and because they allow independent variables to influence WTP in a multiplicative rather than additive manner.

Model Two is a trans-log model, identical to the semi-log specification save for the inclusion of water quality measures (*baseline* and *quality_ch*) as natural logarithms. This form—common in the hedonic modeling literature (Johnston et al. 2001) and illustrated by Johnston et al. (2005) within the meta-analysis literature—shares many advantages of the semi-log functional form, but also incorporates the desirable quality that WTP is constrained to zero when quality change is also equal to zero.

Following standard econometric practice and the “weak structural utility theory” approach to meta-analysis summarized above (Bergstrom and Taylor 2006), the final models are specified based on guidance from theory and prior literature. For example, Arrow et al. (1993) made a fundamental distinction between discrete choice and open-ended payment mechanisms (such as iterative bidding and payment cards). Hence, this distinction is made in the final model (i.e., including the variable *discrete_ch*). Similarly, other “survey methodology” variables in the model were chosen based on theoretical considerations and prior findings in the literature (e.g., voluntary versus mandatory payment vehicles, parametric versus non-parametric, treatment of protest and outlier bids, use of mean versus median WTP). Also included are variables characterizing the scope and scale of the resource change, based on theoretical expectations that such factors should be relevant to welfare estimates.

Few variables were excluded solely because of lack of statistical significance. Individual variables were only excluded if they could not be shown to be statistically significant in any version of the model (restricted or unrestricted), and there was no overriding rationale for retaining the variable in the model. For example, variables distinguishing different *types* of discrete choice instruments (e.g., conjoint versus dichotomous choice) added no significant explanatory power to the model ($p = 0.58$).

It is important to note that although empirical considerations play a role in model development, certain variables were retained in the model for theoretical reasons, even if significance levels were low. Such specification of meta-analysis models using a combination of theoretical guidance and empirical considerations is standard in modeling efforts (Bergstrom and Taylor 2006).

Table G-3 presents results of the total WTP regression model.

Table G-3: Estimated Multilevel Model Results for the Trans-log and Semi-log Total WTP Regression Models: WTP for Aquatic Habitat Improvements

Variable	Trans-log Model		Semi-log Model	
	Parameter Estimate	Standard Error	Parameter Estimate	Standard Error
intercept	5.7109 [3]	0.9352	6.0946 [3]	0.5127
year_indx	-0.08043 [3]	0.01482	-0.06707 [3]	0.01430
discrete	-0.1248	0.2230	-0.1696	0.1735
volunt	-1.3233 [3]	0.1653	-1.3049[3]	0.1661
nonparam	-0.6698[3]	0.1434	-0.6892[3]	0.1164
income	2.698×10^{-6}	3.9×10^{-6}	-1.16×10^{-7}	3.238×10^{-6}
WQI	-0.3275	0.2692	-0.3631[1]	0.2096
outlier_bids	-0.8837[3]	0.2855	-0.7829[3]	0.2164
single_river	-0.4279 [3]	0.1412	-0.2724[2]	0.1222
single_lake	-0.06316	0.2386	-0.1268	0.2223
multiple_river	-1.4752[3]	0.3540	-1.5054[3]	0.3184
regional_fresh	0.1588	0.1505	0.2219[1]	0.1168
salt_pond	0.9849[3]	0.3580	1.1357[3]	0.3142
num_riv_pond	0.1173[3]	0. 02806	0.1145[3]	0.02510
mr	-0.8846[3]	0.1832	-0.7932[3]	0.1343
mp	1.6337[3]	0.2980	1.5168[3]	0.2574
nonusers	-0.4036[3]	0.1314	-0.4451[3]	0.1301
allmult	-0.3728[2]	0.1644	-0.4044[3]	0.1266
nonspec	-0.4042[2]	0.1731	-0.2988	0.1351
quality ch (lnquality_ch in trans-log model)	0.4065[1]	0.1488	0.03208[3]	0.006337
fish_use	-0.3317[2]	0.1291	-0.4480[3]	0.1034
fishplus	0.4432[2]	0.1820	0.4017[3]	0.1217
baseline (lnbase in trans-log model)	0.02610	0.1183	0.005205	0.003739
mail	-0.2013	0.1466	-0.3073[3]	0.1155
lump_sum	0.5569[2]	0.2387	0.7188[3]	0.1873
non_reviewed	-0.2718[1]	0.1625	-0.3744[3]	0.1234
median_wtp	-0.5358[3]	0.1875	-0.4675[2]	0.1898
-2 Log Likelihood	133.9[3]		115.5[2]	
Covariance Factors:				
Study Level (σ_u)	0		0	
Residual (σ_e)	0.1876[3]		0.1599[3]	

[1] Significant at the 0.10 level.
[2] Significant at the 0.05 level.
[3] Significant at the 0.01 level.

G.2.3 Interpretation of Total WTP Regression Analysis Results

Regression results reveal strong systematic elements influencing WTP. The analysis finds both statistically significant and intuitive patterns that influence WTP for surface water quality improvements. In general, the statistical fit of the three estimated equations is good; model results suggest a considerable systematic component of WTP variation that allows forecasting of WTP based on site and study characteristics. Likelihood ratio tests *(Table G-3)* show that model variables are jointly significant at the $p<0.01$ level for the trans-log model and the $p < 0.05$ level for the semi-log model. In both models, the majority of independent variables are statistically significant at $p<0.05$, with most statistically significant at $p<0.01$ (*Table G-3*). Signs of significant parameter estimates generally correspond with intuition, where prior expectations exist. As shown in *Table G-3,* the random effect is statistically insignificant (i.e., study level covariance factors are zero). Considering these factors, the statistical performance of both models compare favorably to prior meta-analyses in the valuation literature.

Despite differences in the functional form of the two models, statistical results are robust across models. In most cases, coefficient magnitudes and standard errors vary to only a small degree. Measures of equation fit are similar, and both models are significant at the $p<0.05$ level or better. Such results mirror those of Johnston et al. (2005), whose earlier meta-analysis of WTP for water quality improvements finds a high degree of robustness to changes in model specification.

The initial discussion emphasizes results of the trans-log model. Although policy implications of both model specifications are nearly identical for moderate to relatively large water quality improvements (e.g., more than 5 percent increase in WQI), the trans-log model provides more accurate WTP estimates when the expected water quality change is very small (e.g., less than 1 percent increase in WQI).

One of the primary means to assess the validity of benefit transfers—and the only one that may be applied in cases wherein the true value for the study site is unknown—is value surface tests (Bergstrom and De Civita 1999). These tests involve assessments of ways in which “different factors may cause values to vary across sites, providing guidance for adjustments needed to make a valid transfer of value estimates from the study site(s) to the policy site” (Bergstrom and De Civita 1999). Following general approaches for such value surface assessments—which generally involve comparisons of empirical patterns found via meta-analysis to theoretical expectations or norms—the Agency concluded that most results of the estimated value surface suggest an appropriate benefit function. Results of these value surface assessments are detailed in the following sections.

Resource Improvement Effects

Seven variables characterize resource improvements and uses; most are of the expected sign. The coefficient on the *quality_ch* variable is positive and statistically significant ($p<0.01$), indicating that larger water quality improvements generate larger WTP. This is an important result, and indicates that WTP is sensitive to the scope of water quality improvements. The estimated model showed that WTP values are not sensitive to the baseline water quality from which water quality change would occur. The estimated parameter on the variable *baseline* representing the baseline water quality from which water quality change would occur is not statistically significant ($p>0.1$). This finding differs from that of Johnston et al. (2005), which shows that WTP for marginal water quality improvements declines as baseline water quality improves. Here, the value is positive but is only significant at $p<0.17$ in the semi-log model. In the trans-log model, this parameter is not statistically significant. The reason for this result is unknown, but may be related to highly valued uses that are associated with larger values on the WQI. For example, increases beginning at higher levels on the WQI may cross thresholds allowing such highly valued uses as swimming and drinking, such that increases at these high-quality levels may be valued more highly than otherwise similar changes at lower baselines, which may not allow such uses. Such thresholds, or other non-convex preference patterns, may lead to unexpected results for the baseline water quality variable (*baseline*).

Both models reveal that water quality changes associated with recreational fishing uses lead to a significant decrease in total WTP values, compared to improvements that do not affect fishing. The variable *fish_use* identifies those studies that specified effects of water quality improvements on recreational fishing (e.g., increase in catch rates). The associated parameter estimate is significant ($p<0.05$) and has the negative sign. In contrast, the variable *fishplus* identifies those studies for which the associated survey identified particularly large gains in fish populations or harvest rates (>50 percent). The positive and statistically significant result ($p<0.05$) indicates that large gains in fish populations or harvests are associated with statistically significant increases in total WTP. Results suggest that while water quality improvements targeting fishing uses may not be valued particularly highly on average, very large resultant improvements in fish populations or harvests are associated with increases in WTP, *ceteris paribus*.

The variables *all_mult* and *nonspec* indicate that water quality improvements affect multiple species (*all_mult*) or unspecified species (*nonspec)*, respectively. The default category from which these variables allow systematic variations in WTP is a focus on particular aquatic species affected by water quality (e.g., shellfish or game fish). The associated coefficients are negative, indicating that WTP is lower when a survey instrument does not specify what aquatic species would be affected (*nonspec)* or when all or multiple species are affected (*all_mult)*. The latter finding seems to be counterintuitive at first. However, when a survey instrument focuses on the effect of water quality on particular species, it is a likely indication that these effects are of significant concern to the affected communities, which typically leads to a higher WTP. That is, this result suggests that WTP is higher when water improvements can be shown to offer targeted benefits to specific, and often high-profile, species groups—as opposed to cases in which improvements benefit an often poorly characterized group of species.

Geographic Region and Scale Effects

Ten binary variables characterize geographic region and scale; seven are statistically significant at $p<0.10$. The default category from which these variables allow systematic variations in WTP is an estuarine waterbody. Also included in this default are a small number of observations addressing national level improvements. Compared to this baseline, WTP associated with rivers is lower (*single_river* and *multiple_river* both have negative and significant values). *Single_lake* has a negative value, but it is not significant. WTP for water quality gains in salt ponds (*salt_pond*) is higher than for estuaries ($p<0.05$). This is not surprising since water quality gains in salt ponds correspond to an increase in the number of acres of shellfish beds.

Of particular importance for the general validity of empirical findings, the model results further suggest that WTP is sensitive to the *number of waterbodies* under consideration and geographic scale of improvement (*regional_fresh*). Of the waterbody categories distinguished above, both rivers and salt ponds allowed variation in numbers of affected waterbodies explicitly described by the survey. This variation is captured by the variable *num_riv_pond* (see *Table G-3).*[29] The associated parameter estimate is statistically significant ($p<0.01$) and indicates that WTP increases with the number of waterbodies considered. The parameter estimate on the r*egional_fresh* variable is positive and significant ($p<0.10$) in the semi-log model, indicating that large-scale regional water quality improvements lead to an increase in WTP. These results, combined with the statistical significance of the water quality change variables noted above, suggest that WTP values (in this case for water quality improvements) are strongly sensitive to

[29] Technically, this variable is the sum of two interaction variables: (1) an interaction between *multiple_river* and the number of waterbodies noted in the survey (0 if unspecified) and (2) an interaction between *salt_pond* and the number of waterbodies noted in the survey (0 if unspecified).

scope, both in terms of the number of waterbodies considered, geographic scale of improvement, and the magnitude of water quality change. In the trans-log model, the r*egional_fresh* variable is positive but not statistically significant (p>0.30).

Finally, the regional indicator variables *mp* and *mult_reg* are statistically significant at p<0.01, suggesting that there are significant differences among WTP estimates from surveys in different geographical regions of the United States. The parameter estimate on the *mult_reg* variable is negative, indicating that WTP for non-local water quality improvements (e.g., out of state) are lower compared to in-state or local resource improvements. This is consistent with prior findings that WTP for water quality improvements declines with the distance from the resource (Bateman et al. 2006). The magnitude of the Pacific Mountain (*mp*) regional effect suggests that spurious or otherwise unexplained effects (e.g., the effect of specific researchers who appear more than once in the data) may drive their overall magnitude. For example, the size of the positive parameter estimate associated with WTP in the Pacific Mountain dummy *(mp)* leads in many cases to relatively large increases in WTP for Pacific Northwest policies. Hence, EPA believes that particular, spurious, or unexplained aspects of studies from this region may have caused the associated parameter estimate to have a larger-than-expected influence on WTP. Although effects of regional dummy variables often escape simple, intuitive characterization, EPA notes that they are often statistically significant in meta-analysis found in the valuation literature. Similar issues are found by Johnston et al. (2005), for example.

Surveyed Populations Effects

Only two variables, *nonusers* and *income*, are used to characterize surveyed populations. In particular, the *nonusers* variable is of substantial policy relevance. The negative and strongly significant (p<.01) parameter estimate indicates that surveys of nonusers only, who by definition have only nonuse values for the resource improvements in question (cf. Freeman 2003, p. 142), generate lower WTP values than surveys that include users, who may have both use and nonuse values. Based on this statistically significant result, it is possible to use this model to estimate nonuse values, interpreted as the mean WTP values estimated by surveys of nonusers only. Such methods, however, may underestimate nonuse values of the general population, if the nonuse values of users exceed those of nonusers (Whitehead and Blomquist 1991a,b).

The *income* parameter estimate is positive in the trans-log model, as expected, but is not statistically significant. Such lack of statistical significance for income parameters is not uncommon in meta-analyses found in the literature (e.g., Johnston et al. 2005).

Study and Methodology Effects

As often found in meta-analyses within the valuation and benefit transfer literature (Navrud and Ready 2007), a variety of study and methodology effects can be shown to influence WTP for water quality improvements. While expected, this does indicate that the methodological approach influences WTP, as argued by Arrow et al. (1993). Of nine variables characterizing study and methodological effects, eight are statistically significant at p<0.10. Among these is the year in which a study was conducted (*year_indx,* a continuous variable), with later studies associated with lower WTP. This is the expected result, as the focus of survey design over time has often been on the reduction of survey biases that would otherwise result in an overstatement of WTP (Arrow et al. 1993).

Model results reveal that voluntary (*voluntary*=1) payment vehicles (i.e., surveys that describe hypothetical payments as voluntary) are associated with reduced WTP estimates. This result counters common intuition and empirical findings that voluntary payment vehicles are associated with overstatements of true WTP (Carson et al. 2000). The reason for this counter-intuitive finding is unknown, but may reflect an unwillingness among respondents to offer large voluntary payments, given the fear that others will free-ride (Johnston et al. 2005). Reduced WTP estimates are also associated with studies applying non-parametric methods to WTP estimation (*nonparam*). Survey elicitation method does not have a strong effect in this model; studies using discrete choice formats have lower WTP values, but this difference is not statistically significant.

Smaller WTP estimates are associated with studies that eliminate or trim outlier bids when estimating WTP (*outlier_bids*=1; p<0.01). Studies that report median WTP (*median_WTP;* p<0.01) have lower WTP values.

Lower WTP is associated with the use of the WQI in the original survey (*WQI*=1). This parameter is, however, significant in the semi-log model only (p<0.1). As is the case with a variety of study design variables, there is no *necessary* expectation with respect to the direction of this effect. Nonetheless, this finding might suggest the capacity of such scales to clarify the specific magnitude and implications of water quality change, and hence (perhaps) reduce methodological misspecification or symbolic biases that might act to systematically inflate estimated WTP.

Survey format variables also have an effect on WTP, as might be expected. *Mail* has a negative and statistically significant coefficient (p<0.01) in the semi-log model, compared to the default of telephone surveys or interviews. This parameter is negative, but not statistically significant (p>0.17) in the trans-log model. It may be possible that the interview and telephone survey format results in larger WTP values either because the respondents are better able to understand the valuation scenario, or because respondents may feel pressure from interviewers to bias their WTP estimates upward. Finally, studies that ask respondents to report an annual payment (as opposed to a *lump_sum* payment) have higher WTP estimates (p<0.05). This likely to reflect the fact that annual payments are regarded as an infinite contribution and may reflect the respondent's uncertainly regarding his future income and budget constraints.

G.2.4 Model Selection

To select the model for estimating benefits of water quality improvements from the regulation, EPA calculated WTP values for a range of WQI changes using both semi-log and trans-log models. In all cases the baseline WQI is set to 50, which approximates the average WQI value across all RF1 reaches in the United States. EPA assigned values to other independent variables corresponding with theory, characteristics of the water resource, and the policy context. *Table 10-11* in *Chapter 10* provides a complete list of values assigned to the remaining independent regressors.

As shown in *Table G-4,* both the semi-log and trans-log models yield similar WTP for water quality changes greater than five points as measured by WQI. However, the semi-log model is not sensitive to very small water quality changes (i.e., changes less than one point on the WQI index). Because the expected water quality changes from the regulation are relatively small, the Agency selected the trans-log specification for estimating benefits from reducing sediment runoff from construction sites, as a more conservative option.

Table G-4: Comparison of WTP for Different Changes in WQI Based on Semi-log and Trans-log Models

Model	20 Point Change in WQI	10 Point Change in WQI	5 Point Change in WQI	1 Point Change in WQI	0.50 Point Change in WQI	0.10 Point Change in WQI	0.01 Point Change in WQI
Trans-log	105.0186	79.2313	59.7760	31.0738	23.4436	12.1869	4.7796
Semi-log	89.0567	64.6167	55.0407	48.4122	47.6419	47.0345	46.8989

Semi-log Error Term = 0.1599.
Trans-log Error Term =0.1876.

G.3 Model Limitations

The validity and reliability of benefit transfer—including that based on meta-analysis—depends on a variety of factors. While benefit transfer can provide valid measures of use and nonuse benefits, tests of its performance have provided mixed results (e.g., Desvousges et al. 1998; Vandenberg et al. 2001; Smith et al. 2002; Shrestha et al. 2007). Nonetheless, benefit transfers are increasingly applied as a core component of benefit cost analyses conducted by EPA and other government agencies (Bergstrom and De Civita 1999; Rosenberger and Phipps 2007). Moreover, Smith et al. (2002, p. 134) argue that "nearly all benefit cost analyses rely on benefit transfers, whether they acknowledge it or not." Given the increasing [or as Smith et al. (2002) might argue, universal] use of benefit transfers, an increasing focus is on the empirical properties of applied transfer methods and models.

Although the statistical performance of the model is good, EPA notes several limitations of the model. These limitations stem largely from information available from the original studies, as well as degrees of freedom and statistical significance. An important factor in any benefit transfer is the ability of the study site or estimated valuation equation to approximate the resource and context under which benefit estimates are desired. As is common, the meta-analysis model presented here provides a close but not perfect match to the context in which values are desired. Although all of the studies used in the meta-analysis valued changes in water quality improvements, many studies did not specify the cause of water quality impairment in the baseline or focused on causes that are different from the pollutant of concern in the regulation (i.e., sediment). Preliminary models, however, suggest no systematic patterns in WTP associated with such factors, at least in the present metadata.

Additional limitations relate to the paucity of demographic variables available for inclusion in the model. The only demographic variable incorporated in the analysis (*income*) was not statistically significant. Moreover, other demographic variables are unavailable.

The estimated model is statistically significant and allows estimation of WTP based on study and site characteristics. However, strictly speaking, model findings are relative to the specific case studies considered, and must be viewed within the context of the 115-observation data set, with all the appropriate caveats. Although this represents a fairly standard-to-large sample size for a meta-analysis in this context (the 45 studies in the analysis gather data from a total of 23,589 respondents), it is relatively small relative to other statistical applications in resource and environmental economics. Model results are also subject to choices regarding functional form and statistical approach, although many of the primary model effects are robust to reasonable changes in functional form and/or statistical methods. The rationale for the specific functional form chosen here (the semi-log form) is detailed above.

As in all cases, results of the meta-analysis are dependent on the sample of studies available for the given resource change (Navrud and Ready 2007), and may be subject to various selection biases if the available literature does not provide a representative, unbiased perspective on welfare estimates associated with resource changes (Rosenberger and Johnston 2007). In this case, however, the Agency took various steps to ameliorate such potential biases, including the incorporation of both peer-reviewed and gray literature to avoid possible publication biases (Rosenberger and Johnston 2007), and the use of a comprehensive literature review in the attempt to avoid—as much as possible—other types of selection biases.

The relatively large (positive) magnitude of the parameter estimate for the Pacific Mountain U.S. regional dummy variable *(mp)* leads EPA to question the appropriate interpretation of this effect. While it is theoretically possible that WTP for water quality changes is substantially higher in the Pacific Northwest (e.g., people who live in this region are outdoor enthusiasts), the magnitude of the effect suggested by the model seems unlikely from an intuitive perspective. As suggested above, it is possible that spurious, unexplained factors influence the magnitude of this parameter in the present model. However, assessments of preliminary model runs suggest that this effect is relatively robust given the present data and selection of variables available. Nonetheless, EPA recommends that the magnitude of the predicted shift in WTP associated with the Pacific Mountain region should be viewed with caution.

Finally, as noted above, there is some controversy over the appropriateness of meta-analysis for benefit transfer. While recognizing this controversy, the Agency emphasizes that the broader literature provides support for the use of various types of meta-analysis for benefit transfer (cf. USEPA 2000b; Bergstrom and Taylor 2006; Shrestha et al. 2007).

Appendix H – Water Quality Index Tables

This appendix presents detailed versions of *Table 10-7* through *Table 10-10,* showing WQI reach mile and percentage improvements by baseline water quality index range, improvement range, and EPA Region for each policy option.

Table H-1: Estimated Water Quality Improvements Under Option 1

EPA Region	Baseline Water Quality	Baseline Scenario		Water Quality Improvements by WQI Change											
				0.01 < ΔWQI < 0.1			0.1 < ΔWQI < 0.5			0.5 < ΔWQI			Total Improved Reaches		
		Reach Miles Modeled	Total Reach Miles in RF1 Network	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles
1	<26	33	37	0	0.00%	0.00%	0	0.00%	0.00%	0	0.00%	0.00%	0	0.00%	0.00%
	26-50	1,343	1,520	77	5.76%	5.09%	0	0.00%	0.00%	9	0.69%	0.61%	87	6.45%	5.70%
	50-70	13,387	15,159	1,481	11.06%	9.77%	31	0.23%	0.21%	20	0.15%	0.13%	1,532	11.44%	10.10%
	>70	1,420	1,608	138	9.69%	8.56%	0	0.00%	0.00%	0	0.00%	0.00%	138	9.69%	8.56%
	Total	**16,182**	**18,324**	**1,696**	**10.48%**	**9.25%**	**31**	**0.19%**	**0.17%**	**29**	**0.18%**	**0.16%**	**1,756**	**10.85%**	**9.58%**
2	<26	82	87	0	0.00%	0.00%	0	0.00%	0.00%	0	0.00%	0.00%	0	0.00%	0.00%
	26-50	1,958	2,084	36	1.82%	1.71%	0	0.00%	0.00%	0	0.00%	0.00%	36	1.82%	1.71%
	50-70	5,507	5,860	177	3.21%	3.01%	0	0.00%	0.00%	0	0.00%	0.00%	177	3.21%	3.01%
	>70	7,592	8,078	202	2.67%	2.51%	0	0.00%	0.00%	0	0.00%	0.00%	202	2.67%	2.51%
	Total	**15,140**	**16,110**	**415**	**2.74%**	**2.57%**	**0**	**0.00%**	**0.00%**	**0**	**0.00%**	**0.00%**	**415**	**2.74%**	**2.57%**
3	<26	1,448	1,684	51	3.49%	3.00%	0	0.00%	0.00%	0	0.00%	0.00%	51	3.49%	3.00%
	26-50	12,552	14,599	645	5.14%	4.42%	21	0.17%	0.14%	0	0.00%	0.00%	666	5.30%	4.56%
	50-70	13,006	15,127	780	5.99%	5.15%	93	0.71%	0.61%	0	0.00%	0.00%	872	6.71%	5.77%
	>70	1,898	2,207	64	3.39%	2.91%	0	0.00%	0.00%	0	0.00%	0.00%	64	3.39%	2.91%
	Total	**28,904**	**33,617**	**1,539**	**5.33%**	**4.58%**	**114**	**0.39%**	**0.34%**	**0**	**0.00%**	**0.00%**	**1,653**	**5.72%**	**4.92%**
4	<26	714	746	135	18.93%	18.11%	0	0.00%	0.00%	0	0.00%	0.00%	135	18.93%	18.11%
	26-50	32,338	33,801	7,903	24.44%	23.38%	1,003	3.10%	2.97%	154	0.48%	0.45%	9,060	28.02%	26.80%
	50-70	38,191	39,918	12,232	32.03%	30.64%	1,696	4.44%	4.25%	282	0.74%	0.71%	14,210	37.21%	35.60%
	>70	19,192	20,060	5,939	30.95%	29.61%	473	2.46%	2.36%	94	0.49%	0.47%	6,506	33.90%	32.43%
	Total	**90,435**	**94,525**	**26,210**	**28.98%**	**27.73%**	**3,172**	**3.51%**	**3.36%**	**529**	**0.59%**	**0.56%**	**29,911**	**33.07%**	**31.64%**
5	<26	4,175	4,375	0	0.00%	0.00%	0	0.00%	0.00%	0	0.00%	0.00%	0	0.00%	0.00%
	26-50	43,315	45,386	899	2.08%	1.98%	33	0.08%	0.07%	4	0.01%	0.01%	937	2.16%	2.06%
	50-70	18,508	19,393	1,859	10.04%	9.58%	99	0.53%	0.51%	0	0.00%	0.00%	1,958	10.58%	10.09%
	>70	2,287	2,396	173	7.56%	7.22%	0	0.00%	0.00%	0	0.00%	0.00%	173	7.56%	7.22%
	Total	**68,285**	**71,550**	**2,931**	**4.29%**	**4.10%**	**132**	**0.19%**	**0.18%**	**4**	**0.01%**	**0.01%**	**3,067**	**4.49%**	**4.29%**

Table H-1: Estimated Water Quality Improvements Under Option 1

EPA Region	Baseline Water Quality	Baseline Scenario		Water Quality Improvements by WQI Change											
				0.01 < ΔWQI < 0.1			0.1 < ΔWQI < 0.5			0.5 < ΔWQI			Total Improved Reaches		
		Reach Miles Modeled	Total Reach Miles in RF1 Network	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles
6	<26	931	966	101	10.82%	10.43%	7	0.71%	0.68%	13	1.43%	1.38%	121	12.96%	12.49%
	26-50	61,473	63,789	7,303	11.88%	11.45%	995	1.62%	1.56%	440	0.72%	0.69%	8,738	14.21%	13.70%
	50-70	26,731	27,738	8,464	31.66%	30.52%	2,419	9.05%	8.72%	587	2.19%	2.12%	11,470	42.91%	41.35%
	>70	5,963	6,188	2,034	34.10%	32.86%	805	13.51%	13.02%	187	3.14%	3.02%	3,026	50.74%	48.90%
	Total	**95,098**	**98,681**	**17,902**	**18.82%**	**18.14%**	**4,227**	**4.44%**	**4.28%**	**1,227**	**1.29%**	**1.24%**	**23,355**	**24.56%**	**23.67%**
7	<26	10,877	10,877	13	0.12%	0.12%	0	0.00%	0.00%	11	0.10%	0.10%	24	0.22%	0.22%
	26-50	43,158	43,158	2,096	4.86%	4.86%	212	0.49%	0.49%	132	0.31%	0.31%	2,439	5.65%	5.65%
	50-70	6,199	6,199	1,933	31.18%	31.18%	328	5.29%	5.29%	26	0.41%	0.41%	2,286	36.88%	36.88%
	>70	675	675	155	22.95%	22.95%	22	3.28%	3.28%	0	0.00%	0.00%	177	26.22%	26.22%
	Total	**60,909**	**60,909**	**4,196**	**6.89%**	**6.89%**	**562**	**0.92%**	**0.92%**	**168**	**0.28%**	**0.28%**	**4,926**	**8.09%**	**8.09%**
8	<26	6,557	6,557	0	0.00%	0.00%	0	0.00%	0.00%	68	1.04%	1.04%	68	1.04%	1.04%
	26-50	66,701	66,701	29	0.04%	0.04%	28	0.04%	0.04%	190	0.28%	0.28%	246	0.37%	0.37%
	50-70	32,368	32,368	363	1.12%	1.12%	36	0.11%	0.11%	100	0.31%	0.31%	500	1.54%	1.54%
	>70	24,685	24,685	103	0.42%	0.42%	0	0.00%	0.00%	4	0.02%	0.02%	107	0.43%	0.43%
	Total	**130,311**	**130,311**	**495**	**0.38%**	**0.38%**	**64**	**0.05%**	**0.05%**	**362**	**0.28%**	**0.28%**	**921**	**0.71%**	**0.71%**
9	<26	10,282	10,712	128	1.24%	1.19%	0	0.00%	0.00%	7	0.07%	0.07%	135	1.32%	1.26%
	26-50	39,696	41,353	1,170	2.95%	2.83%	128	0.32%	0.31%	144	0.36%	0.35%	1,442	3.63%	3.49%
	50-70	3,946	4,111	62	1.57%	1.51%	0	0.00%	0.00%	0	0.00%	0.00%	62	1.57%	1.51%
	>70	304	316	0	0.00%	0.00%	6	2.06%	1.97%	0	0.00%	0.00%	6	2.06%	1.97%
	Total	**54,228**	**56,492**	**1,360**	**2.51%**	**2.41%**	**134**	**0.25%**	**0.24%**	**151**	**0.28%**	**0.27%**	**1,646**	**3.03%**	**2.91%**

Table H-1: Estimated Water Quality Improvements Under Option 1

EPA Region	Baseline Water Quality	Baseline Scenario		Water Quality Improvements by WQI Change											
				0.01 < ΔWQI < 0.1			0.1 < ΔWQI < 0.5			0.5 < ΔWQI			Total Improved Reaches		
		Reach Miles Modeled	Total Reach Miles in RF1 Network	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles
10	<26	39	40	0	0.00%	0.00%	0	0.00%	0.00%	0	0.00%	0.00%	0	0.00%	0.00%
	26-50	13,116	13,373	263	2.01%	1.97%	189	1.44%	1.41%	28	0.21%	0.21%	480	3.66%	3.59%
	50-70	24,190	24,664	1,043	4.31%	4.23%	410	1.69%	1.66%	213	0.88%	0.86%	1,666	6.89%	6.75%
	>70	30,844	31,448	2,235	7.24%	7.11%	723	2.34%	2.30%	301	0.98%	0.96%	3,259	10.57%	10.36%
	Total	**68,189**	**69,524**	**3,541**	**5.19%**	**5.09%**	**1,322**	**1.94%**	**1.90%**	**542**	**0.79%**	**0.78%**	**5,404**	**7.93%**	**7.77%**
Nation	<26	35,137	36,080	427	1.22%	1.18%	7	0.02%	0.02%	99	0.28%	0.27%	533	1.52%	1.48%
	26-50	315,650	325,764	20,421	6.47%	6.27%	2,609	0.83%	0.80%	1,100	0.35%	0.34%	24,131	7.64%	7.41%
	50-70	182,033	190,537	28,393	15.60%	14.90%	5,112	2.81%	2.68%	1,227	0.67%	0.64%	34,732	19.08%	18.23%
	>70	94,859	97,662	11,042	11.64%	11.31%	2,030	2.14%	2.08%	586	0.62%	0.60%	13,658	14.40%	13.99%
	Total	**627,679**	**650,043**	**60,285**	**9.60%**	**9.27%**	**9,757**	**1.55%**	**1.50%**	**3,012**	**0.48%**	**0.46%**	**73,054**	**11.64%**	**11.24%**

Table H-2: Estimated Water Quality Improvements Under Option 2

EPA Region	Baseline Water Quality	Baseline Scenario		Water Quality Improvements by WQI Change											
				0.01 < ΔWQI < 0.1			0.1 < ΔWQI < 0.5			0.5 < ΔWQI			Total Improved Reaches		
		Reach Miles Modeled	Total Reach Miles in RF1 Network	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles
1	<26	33	37	3	8.79%	7.76%	0	0.00%	0.00%	0	0.00%	0.00%	3	8.79%	7.76%
	26-50	1,343	1,520	176	13.10%	11.57%	5	0.40%	0.35%	9	0.69%	0.61%	191	14.19%	12.53%
	50-70	13,387	15,159	2,606	19.46%	17.19%	229	1.71%	1.51%	32	0.24%	0.21%	2,866	21.41%	18.91%
	>70	1,420	1,608	250	17.63%	15.57%	0	0.00%	0.00%	0	0.00%	0.00%	250	17.63%	15.57%
	Total	**16,182**	**18,324**	**3,035**	**18.75%**	**16.56%**	**235**	**1.45%**	**1.28%**	**41**	**0.25%**	**0.22%**	**3,310**	**20.46%**	**18.06%**
2	<26	82	87	0	0.00%	0.00%	0	0.00%	0.00%	0	0.00%	0.00%	0	0.00%	0.00%
	26-50	1,958	2,084	90	4.59%	4.31%	0	0.00%	0.00%	0	0.00%	0.00%	90	4.59%	4.31%
	50-70	5,507	5,860	766	13.91%	13.07%	0	0.00%	0.00%	0	0.00%	0.00%	766	13.91%	13.07%
	>70	7,592	8,078	671	8.84%	8.31%	5	0.07%	0.06%	0	0.00%	0.00%	676	8.91%	8.37%
	Total	**15,140**	**16,110**	**1,527**	**10.09%**	**9.48%**	**5**	**0.03%**	**0.03%**	**0**	**0.00%**	**0.00%**	**1,532**	**10.12%**	**9.51%**
3	<26	1,448	1,684	144	9.93%	8.54%	0	0.00%	0.00%	0	0.00%	0.00%	144	9.93%	8.54%
	26-50	12,552	14,599	1,907	15.19%	13.06%	28	0.22%	0.19%	0	0.00%	0.00%	1,934	15.41%	13.25%
	50-70	13,006	15,127	2,174	16.72%	14.37%	80	0.62%	0.53%	30	0.23%	0.20%	2,285	17.56%	15.10%
	>70	1,898	2,207	258	13.59%	11.69%	0	0.00%	0.00%	0	0.00%	0.00%	258	13.59%	11.69%
	Total	**28,904**	**33,617**	**4,483**	**15.51%**	**13.34%**	**108**	**0.37%**	**0.32%**	**30**	**0.10%**	**0.09%**	**4,621**	**15.99%**	**13.74%**
4	<26	714	746	221	30.97%	29.63%	13	1.89%	1.80%	0	0.00%	0.00%	234	32.85%	31.43%
	26-50	32,338	33,801	11,822	36.56%	34.97%	1,959	6.06%	5.80%	458	1.42%	1.36%	14,239	44.03%	42.13%
	50-70	38,191	39,918	17,431	45.64%	43.67%	3,450	9.03%	8.64%	639	1.67%	1.60%	21,520	56.35%	53.91%
	>70	19,192	20,060	8,462	44.09%	42.18%	1,171	6.10%	5.84%	191	0.99%	0.95%	9,824	51.19%	48.97%
	Total	**90,435**	**94,525**	**37,936**	**41.95%**	**40.13%**	**6,594**	**7.29%**	**6.98%**	**1,288**	**1.42%**	**1.36%**	**45,817**	**50.66%**	**48.47%**
5	<26	4,175	4,375	31	0.75%	0.71%	0	0.00%	0.00%	0	0.00%	0.00%	31	0.75%	0.71%
	26-50	43,315	45,386	2,136	4.93%	4.71%	89	0.21%	0.20%	25	0.06%	0.05%	2,250	5.19%	4.96%
	50-70	18,508	19,393	3,268	17.66%	16.85%	320	1.73%	1.65%	60	0.33%	0.31%	3,648	19.71%	18.81%
	>70	2,287	2,396	454	19.84%	18.94%	0	0.00%	0.00%	0	0.00%	0.00%	454	19.84%	18.94%
	Total	**68,285**	**71,550**	**5,889**	**8.62%**	**8.23%**	**409**	**0.60%**	**0.57%**	**85**	**0.12%**	**0.12%**	**6,383**	**9.35%**	**8.92%**

Table H-2: Estimated Water Quality Improvements Under Option 2

EPA Region	Baseline Water Quality	Baseline Scenario		Water Quality Improvements by WQI Change											
				0.01 < ΔWQI < 0.1			0.1 < ΔWQI < 0.5			0.5 < ΔWQI			Total Improved Reaches		
		Reach Miles Modeled	Total Reach Miles in RF1 Network	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles
6	<26	931	966	138	14.86%	14.32%	0	0.00%	0.00%	46	4.93%	4.76%	184	19.79%	19.07%
	26-50	61,473	63,789	11,393	18.53%	17.86%	1,896	3.08%	2.97%	825	1.34%	1.29%	14,114	22.96%	22.13%
	50-70	26,731	27,738	8,353	31.25%	30.11%	4,035	15.09%	14.55%	1,488	5.57%	5.36%	13,876	51.91%	50.02%
	>70	5,963	6,188	1,558	26.13%	25.18%	1,255	21.04%	20.27%	554	9.29%	8.95%	3,366	56.45%	54.40%
	Total	**95,098**	**98,681**	**21,442**	**22.55%**	**21.73%**	**7,185**	**7.56%**	**7.28%**	**2,912**	**3.06%**	**2.95%**	**31,540**	**33.17%**	**31.96%**
7	<26	10,877	10,877	47	0.43%	0.43%	0	0.00%	0.00%	11	0.10%	0.10%	57	0.53%	0.53%
	26-50	43,158	43,158	3,854	8.93%	8.93%	420	0.97%	0.97%	179	0.42%	0.42%	4,453	10.32%	10.32%
	50-70	6,199	6,199	2,546	41.07%	41.07%	557	8.99%	8.99%	71	1.15%	1.15%	3,174	51.21%	51.21%
	>70	675	675	273	40.40%	40.40%	22	3.28%	3.28%	0	0.00%	0.00%	295	43.68%	43.68%
	Total	**60,909**	**60,909**	**6,719**	**11.03%**	**11.03%**	**1,000**	**1.64%**	**1.64%**	**261**	**0.43%**	**0.43%**	**7,980**	**13.10%**	**13.10%**
8	<26	6,557	6,557	13	0.20%	0.20%	0	0.00%	0.00%	68	1.04%	1.04%	81	1.24%	1.24%
	26-50	66,701	66,701	253	0.38%	0.38%	39	0.06%	0.06%	190	0.28%	0.28%	482	0.72%	0.72%
	50-70	32,368	32,368	773	2.39%	2.39%	51	0.16%	0.16%	105	0.32%	0.32%	928	2.87%	2.87%
	>70	24,685	24,685	326	1.32%	1.32%	0	0.00%	0.00%	4	0.02%	0.02%	330	1.34%	1.34%
	Total	**130,311**	**130,311**	**1,365**	**1.05%**	**1.05%**	**90**	**0.07%**	**0.07%**	**367**	**0.28%**	**0.28%**	**1,821**	**1.40%**	**1.40%**
9	<26	10,282	10,712	239	2.33%	2.23%	0	0.00%	0.00%	7	0.07%	0.07%	246	2.40%	2.30%
	26-50	39,696	41,353	1,741	4.39%	4.21%	213	0.54%	0.51%	179	0.45%	0.43%	2,133	5.37%	5.16%
	50-70	3,946	4,111	102	2.58%	2.48%	0	0.00%	0.00%	0	0.00%	0.00%	102	2.58%	2.48%
	>70	304	316	6	2.00%	1.92%	6	2.06%	1.97%	0	0.00%	0.00%	12	4.06%	3.89%
	Total	**54,228**	**56,492**	**2,088**	**3.85%**	**3.70%**	**219**	**0.40%**	**0.39%**	**186**	**0.34%**	**0.33%**	**2,493**	**4.60%**	**4.41%**

Table H-2: Estimated Water Quality Improvements Under Option 2

EPA Region	Baseline Water Quality	Baseline Scenario		Water Quality Improvements by WQI Change											
				0.01 < ΔWQI < 0.1			0.1 < ΔWQI < 0.5			0.5 < ΔWQI			Total Improved Reaches		
		Reach Miles Modeled	Total Reach Miles in RF1 Network	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles
10	<26	39	40	0	0.00%	0.00%	0	0.00%	0.00%	0	0.00%	0.00%	0	0.00%	0.00%
	26-50	13,116	13,373	220	1.68%	1.64%	257	1.96%	1.92%	117	0.89%	0.87%	594	4.53%	4.44%
	50-70	24,190	24,664	1,538	6.36%	6.24%	250	1.03%	1.01%	420	1.74%	1.70%	2,208	9.13%	8.95%
	>70	30,844	31,448	2,537	8.23%	8.07%	1,084	3.52%	3.45%	508	1.65%	1.62%	4,129	13.39%	13.13%
	Total	**68,189**	**69,524**	**4,295**	**6.30%**	**6.18%**	**1,592**	**2.33%**	**2.29%**	**1,044**	**1.53%**	**1.50%**	**6,931**	**10.16%**	**9.97%**
Nation	<26	35,137	36,080	836	2.38%	2.32%	13	0.04%	0.04%	132	0.38%	0.37%	982	2.79%	2.72%
	26-50	315,650	325,764	33,591	10.64%	10.31%	4,907	1.55%	1.51%	1,981	0.63%	0.61%	40,479	12.82%	12.43%
	50-70	182,033	190,537	39,557	21.73%	20.76%	8,972	4.93%	4.71%	2,844	1.56%	1.49%	51,373	28.22%	26.96%
	>70	94,859	97,662	14,795	15.60%	15.15%	3,543	3.74%	3.63%	1,257	1.32%	1.29%	19,595	20.66%	20.06%
	Total	**627,679**	**650,043**	**88,779**	**14.14%**	**13.66%**	**17,436**	**2.78%**	**2.68%**	**6,214**	**0.99%**	**0.96%**	**112,429**	**17.91%**	**17.30%**

Table H-3: Estimated Water Quality Improvements Under Option 3

EPA Region	Baseline Water Quality	Baseline Scenario		Water Quality Improvements by WQI Change											
				0.01 < ΔWQI < 0.1			0.1 < ΔWQI < 0.5			0.5 < ΔWQI			Total Improved Reaches		
		Reach Miles Modeled	Total Reach Miles in RF1 Network	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles
1	<26	33	37	3	8.79%	7.76%	0	0.00%	0.00%	0	0.00%	0.00%	3	8.79%	7.76%
	26-50	1,343	1,520	223	16.60%	14.66%	5	0.40%	0.35%	9	0.69%	0.61%	238	17.70%	15.63%
	50-70	13,387	15,159	3,142	23.47%	20.73%	295	2.21%	1.95%	32	0.24%	0.21%	3,469	25.91%	22.88%
	>70	1,420	1,608	349	24.60%	21.73%	0	0.00%	0.00%	0	0.00%	0.00%	349	24.60%	21.73%
	Total	**16,182**	**18,324**	**3,717**	**22.97%**	**20.29%**	**301**	**1.86%**	**1.64%**	**41**	**0.25%**	**0.22%**	**4,059**	**25.08%**	**22.15%**
2	<26	82	87	0	0.00%	0.00%	0	0.00%	0.00%	0	0.00%	0.00%	0	0.00%	0.00%
	26-50	1,958	2,084	155	7.90%	7.42%	0	0.00%	0.00%	0	0.00%	0.00%	155	7.90%	7.42%
	50-70	5,507	5,860	1,026	18.63%	17.51%	0	0.00%	0.00%	0	0.00%	0.00%	1,026	18.63%	17.51%
	>70	7,592	8,078	961	12.66%	11.90%	5	0.07%	0.06%	0	0.00%	0.00%	966	12.72%	11.96%
	Total	**15,140**	**16,110**	**2,142**	**14.15%**	**13.29%**	**5**	**0.03%**	**0.03%**	**0**	**0.00%**	**0.00%**	**2,147**	**14.18%**	**13.33%**
3	<26	1,448	1,684	155	10.74%	9.23%	0	0.00%	0.00%	0	0.00%	0.00%	155	10.74%	9.23%
	26-50	12,552	14,599	2,449	19.51%	16.77%	35	0.28%	0.24%	0	0.00%	0.00%	2,483	19.78%	17.01%
	50-70	13,006	15,127	3,135	24.10%	20.72%	121	0.93%	0.80%	30	0.23%	0.20%	3,285	25.26%	21.72%
	>70	1,898	2,207	346	18.21%	15.66%	7	0.38%	0.33%	0	0.00%	0.00%	353	18.59%	15.99%
	Total	**28,904**	**33,617**	**6,084**	**21.05%**	**18.10%**	**163**	**0.56%**	**0.48%**	**30**	**0.10%**	**0.09%**	**6,277**	**21.72%**	**18.67%**
4	<26	714	746	193	27.00%	25.83%	44	6.18%	5.92%	0	0.00%	0.00%	237	33.18%	31.75%
	26-50	32,338	33,801	12,663	39.16%	37.46%	2,401	7.43%	7.10%	618	1.91%	1.83%	15,683	48.50%	46.40%
	50-70	38,191	39,918	18,276	47.85%	45.78%	4,249	11.12%	10.64%	771	2.02%	1.93%	23,296	61.00%	58.36%
	>70	19,192	20,060	8,960	46.68%	44.66%	1,549	8.07%	7.72%	220	1.15%	1.10%	10,729	55.90%	53.48%
	Total	**90,435**	**94,525**	**40,092**	**44.33%**	**42.41%**	**8,244**	**9.12%**	**8.72%**	**1,610**	**1.78%**	**1.70%**	**49,945**	**55.23%**	**52.84%**

Table H-3: Estimated Water Quality Improvements Under Option 3

EPA Region	Baseline Water Quality	Baseline Scenario		Water Quality Improvements by WQI Change											
				0.01 < ΔWQI < 0.1			0.1 < ΔWQI < 0.5			0.5 < ΔWQI			Total Improved Reaches		
		Reach Miles Modeled	Total Reach Miles in RF1 Network	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles
5	<26	4,175	4,375	58	1.39%	1.32%	0	0.00%	0.00%	0	0.00%	0.00%	58	1.39%	1.32%
	26-50	43,315	45,386	2,595	5.99%	5.72%	137	0.32%	0.30%	25	0.06%	0.05%	2,757	6.36%	6.07%
	50-70	18,508	19,393	3,983	21.52%	20.54%	493	2.67%	2.54%	60	0.33%	0.31%	4,537	24.51%	23.40%
	>70	2,287	2,396	592	25.88%	24.70%	0	0.00%	0.00%	0	0.00%	0.00%	592	25.88%	24.70%
	Total	**68,285**	**71,550**	**7,228**	**10.59%**	**10.10%**	**630**	**0.92%**	**0.88%**	**85**	**0.12%**	**0.12%**	**7,943**	**11.63%**	**11.10%**
6	<26	931	966	185	19.90%	19.18%	0	0.00%	0.00%	66	7.11%	6.86%	252	27.02%	26.04%
	26-50	61,473	63,789	13,385	21.77%	20.98%	2,174	3.54%	3.41%	1,010	1.64%	1.58%	16,569	26.95%	25.97%
	50-70	26,731	27,738	8,577	32.09%	30.92%	4,786	17.90%	17.25%	1,741	6.51%	6.28%	15,104	56.50%	54.45%
	>70	5,963	6,188	1,532	25.69%	24.76%	1,268	21.27%	20.50%	671	11.25%	10.84%	3,471	58.21%	56.09%
	Total	**95,098**	**98,681**	**23,679**	**24.90%**	**24.00%**	**8,228**	**8.65%**	**8.34%**	**3,487**	**3.67%**	**3.53%**	**35,395**	**37.22%**	**35.87%**
7	<26	10,877	10,877	34	0.31%	0.31%	13	0.12%	0.12%	11	0.10%	0.10%	57	0.53%	0.53%
	26-50	43,158	43,158	4,404	10.21%	10.21%	527	1.22%	1.22%	233	0.54%	0.54%	5,164	11.96%	11.96%
	50-70	6,199	6,199	2,718	43.85%	43.85%	732	11.81%	11.81%	76	1.22%	1.22%	3,526	56.88%	56.88%
	>70	675	675	317	46.97%	46.97%	28	4.13%	4.13%	0	0.00%	0.00%	345	51.10%	51.10%
	Total	**60,909**	**60,909**	**7,473**	**12.27%**	**12.27%**	**1,300**	**2.13%**	**2.13%**	**319**	**0.52%**	**0.52%**	**9,092**	**14.93%**	**14.93%**
8	<26	6,557	6,557	26	0.39%	0.39%	0	0.00%	0.00%	68	1.04%	1.04%	94	1.43%	1.43%
	26-50	66,701	66,701	1,348	2.02%	2.02%	39	0.06%	0.06%	190	0.28%	0.28%	1,577	2.36%	2.36%
	50-70	32,368	32,368	1,826	5.64%	5.64%	72	0.22%	0.22%	114	0.35%	0.35%	2,011	6.21%	6.21%
	>70	24,685	24,685	477	1.93%	1.93%	0	0.00%	0.00%	4	0.02%	0.02%	482	1.95%	1.95%
	Total	**130,311**	**130,311**	**3,677**	**2.82%**	**2.82%**	**111**	**0.09%**	**0.09%**	**376**	**0.29%**	**0.29%**	**4,164**	**3.20%**	**3.20%**
9	<26	10,282	10,712	295	2.87%	2.75%	0	0.00%	0.00%	7	0.07%	0.07%	302	2.94%	2.82%
	26-50	39,696	41,353	2,334	5.88%	5.64%	275	0.69%	0.67%	215	0.54%	0.52%	2,825	7.12%	6.83%
	50-70	3,946	4,111	199	5.03%	4.83%	0	0.00%	0.00%	0	0.00%	0.00%	199	5.03%	4.83%
	>70	304	316	6	2.00%	1.92%	6	2.06%	1.97%	0	0.00%	0.00%	12	4.06%	3.89%
	Total	**54,228**	**56,492**	**2,834**	**5.23%**	**5.02%**	**282**	**0.52%**	**0.50%**	**222**	**0.41%**	**0.39%**	**3,338**	**6.16%**	**5.91%**

Table H-3: Estimated Water Quality Improvements Under Option 3

EPA Region	Baseline Water Quality	Baseline Scenario		Water Quality Improvements by WQI Change											
				0.01 < ΔWQI < 0.1			0.1 < ΔWQI < 0.5			0.5 < ΔWQI			Total Improved Reaches		
		Reach Miles Modeled	Total Reach Miles in RF1 Network	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles
10	<26	39	40	0	0.00%	0.00%	0	0.00%	0.00%	0	0.00%	0.00%	0	0.00%	0.00%
	26-50	13,116	13,373	224	1.70%	1.67%	314	2.39%	2.35%	117	0.89%	0.87%	654	4.99%	4.89%
	50-70	24,190	24,664	1,616	6.68%	6.55%	358	1.48%	1.45%	446	1.85%	1.81%	2,420	10.01%	9.81%
	>70	30,844	31,448	2,566	8.32%	8.16%	1,140	3.69%	3.62%	607	1.97%	1.93%	4,312	13.98%	13.71%
	Total	**68,189**	**69,524**	**4,405**	**6.46%**	**6.34%**	**1,812**	**2.66%**	**2.61%**	**1,170**	**1.72%**	**1.68%**	**7,387**	**10.83%**	**10.63%**
Nation	<26	35,137	36,080	949	2.70%	2.63%	57	0.16%	0.16%	152	0.43%	0.42%	1,158	3.30%	3.21%
	26-50	315,650	325,764	39,780	12.60%	12.21%	5,908	1.87%	1.81%	2,416	0.77%	0.74%	48,104	15.24%	14.77%
	50-70	182,033	190,537	44,498	24.44%	23.35%	11,106	6.10%	5.83%	3,270	1.80%	1.72%	58,874	32.34%	30.90%
	>70	94,859	97,662	16,106	16.98%	16.49%	4,004	4.22%	4.10%	1,502	1.58%	1.54%	21,612	22.78%	22.13%
	Total	**627,679**	**650,043**	**101,332**	**16.14%**	**15.59%**	**21,075**	**3.36%**	**3.24%**	**7,340**	**1.17%**	**1.13%**	**129,747**	**20.67%**	**19.96%**

Table H-4: Estimated Water Quality Improvements Under Option 4

EPA Region	Baseline Water Quality	Baseline Scenario		Water Quality Improvements by WQI Change											
				0.01 < ΔWQI < 0.1			0.1 < ΔWQI < 0.5			0.5 < ΔWQI			Total Improved Reaches		
		Reach Miles Modeled	Total Reach Miles in RF1 Network	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles
1	<26	33	37	0	0.00%	0.00%	0	0.00%	0.00%	0	0.00%	0.00%	0	0.00%	0.00%
	26-50	1,343	1,520	131	9.76%	8.62%	0	0.00%	0.00%	9	0.69%	0.61%	140	10.45%	9.23%
	50-70	13,387	15,159	1,982	14.81%	13.08%	156	1.16%	1.03%	20	0.15%	0.13%	2,158	16.12%	14.24%
	>70	1,420	1,608	224	15.80%	13.95%	0	0.00%	0.00%	0	0.00%	0.00%	224	15.80%	13.95%
	Total	**16,182**	**18,324**	**2,338**	**14.45%**	**12.76%**	**156**	**0.96%**	**0.85%**	**29**	**0.18%**	**0.16%**	**2,522**	**15.59%**	**13.77%**
2	<26	82	87	0	0.00%	0.00%	0	0.00%	0.00%	0	0.00%	0.00%	0	0.00%	0.00%
	26-50	1,958	2,084	90	4.59%	4.31%	0	0.00%	0.00%	0	0.00%	0.00%	90	4.59%	4.31%
	50-70	5,507	5,860	713	12.95%	12.17%	0	0.00%	0.00%	0	0.00%	0.00%	713	12.95%	12.17%
	>70	7,592	8,078	630	8.30%	7.80%	5	0.07%	0.06%	0	0.00%	0.00%	635	8.37%	7.87%
	Total	**15,140**	**16,110**	**1,433**	**9.47%**	**8.90%**	**5**	**0.03%**	**0.03%**	**0**	**0.00%**	**0.00%**	**1,438**	**9.50%**	**8.93%**
3	<26	1,448	1,684	144	9.93%	8.54%	0	0.00%	0.00%	0	0.00%	0.00%	144	9.93%	8.54%
	26-50	12,552	14,599	2,031	16.18%	13.91%	28	0.22%	0.19%	0	0.00%	0.00%	2,058	16.40%	14.10%
	50-70	13,006	15,127	2,229	17.14%	14.74%	80	0.62%	0.53%	30	0.23%	0.20%	2,339	17.98%	15.46%
	>70	1,898	2,207	217	11.41%	9.81%	0	0.00%	0.00%	0	0.00%	0.00%	217	11.41%	9.81%
	Total	**28,904**	**33,617**	**4,620**	**15.98%**	**13.74%**	**108**	**0.37%**	**0.32%**	**30**	**0.10%**	**0.09%**	**4,758**	**16.46%**	**14.15%**
4	<26	714	746	234	32.85%	31.43%	0	0.00%	0.00%	0	0.00%	0.00%	234	32.85%	31.43%
	26-50	32,338	33,801	12,208	37.75%	36.12%	2,084	6.44%	6.16%	531	1.64%	1.57%	14,823	45.84%	43.86%
	50-70	38,191	39,918	17,484	45.78%	43.80%	3,785	9.91%	9.48%	714	1.87%	1.79%	21,982	57.56%	55.07%
	>70	19,192	20,060	8,581	44.71%	42.77%	1,290	6.72%	6.43%	220	1.15%	1.10%	10,091	52.58%	50.30%
	Total	**90,435**	**94,525**	**38,507**	**42.58%**	**40.74%**	**7,159**	**7.92%**	**7.57%**	**1,465**	**1.62%**	**1.55%**	**47,130**	**52.12%**	**49.86%**
5	<26	4,175	4,375	58	1.39%	1.32%	0	0.00%	0.00%	0	0.00%	0.00%	58	1.39%	1.32%
	26-50	43,315	45,386	2,184	5.04%	4.81%	68	0.16%	0.15%	25	0.06%	0.05%	2,277	5.26%	5.02%
	50-70	18,508	19,393	3,242	17.52%	16.72%	308	1.67%	1.59%	60	0.33%	0.31%	3,611	19.51%	18.62%
	>70	2,287	2,396	495	21.65%	20.66%	0	0.00%	0.00%	0	0.00%	0.00%	495	21.65%	20.66%
	Total	**68,285**	**71,550**	**5,979**	**8.76%**	**8.36%**	**377**	**0.55%**	**0.53%**	**85**	**0.12%**	**0.12%**	**6,441**	**9.43%**	**9.00%**

Table H-4: Estimated Water Quality Improvements Under Option 4

EPA Region	Baseline Water Quality	Baseline Scenario		Water Quality Improvements by WQI Change											
				0.01 < ΔWQI < 0.1			0.1 < ΔWQI < 0.5			0.5 < ΔWQI			Total Improved Reaches		
		Reach Miles Modeled	Total Reach Miles in RF1 Network	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles
6	<26	931	966	185	19.90%	19.18%	0	0.00%	0.00%	46	4.93%	4.76%	231	24.84%	23.94%
	26-50	61,473	63,789	11,913	19.38%	18.68%	2,085	3.39%	3.27%	945	1.54%	1.48%	14,943	24.31%	23.43%
	50-70	26,731	27,738	8,090	30.26%	29.17%	4,456	16.67%	16.07%	1,698	6.35%	6.12%	14,245	53.29%	51.35%
	>70	5,963	6,188	1,504	25.22%	24.31%	1,278	21.44%	20.66%	624	10.46%	10.08%	3,406	57.12%	55.05%
	Total	**95,098**	**98,681**	**21,693**	**22.81%**	**21.98%**	**7,819**	**8.22%**	**7.92%**	**3,313**	**3.48%**	**3.36%**	**32,825**	**34.52%**	**33.26%**
7	<26	10,877	10,877	34	0.31%	0.31%	13	0.12%	0.12%	11	0.10%	0.10%	57	0.53%	0.53%
	26-50	43,158	43,158	3,922	9.09%	9.09%	524	1.22%	1.22%	193	0.45%	0.45%	4,640	10.75%	10.75%
	50-70	6,199	6,199	2,604	42.01%	42.01%	639	10.31%	10.31%	76	1.22%	1.22%	3,319	53.55%	53.55%
	>70	675	675	300	44.40%	44.40%	28	4.13%	4.13%	0	0.00%	0.00%	328	48.53%	48.53%
	Total	**60,909**	**60,909**	**6,860**	**11.26%**	**11.26%**	**1,205**	**1.98%**	**1.98%**	**280**	**0.46%**	**0.46%**	**8,345**	**13.70%**	**13.70%**
8	<26	6,557	6,557	26	0.39%	0.39%	0	0.00%	0.00%	68	1.04%	1.04%	94	1.43%	1.43%
	26-50	66,701	66,701	250	0.37%	0.37%	39	0.06%	0.06%	190	0.28%	0.28%	479	0.72%	0.72%
	50-70	32,368	32,368	711	2.20%	2.20%	51	0.16%	0.16%	105	0.32%	0.32%	866	2.68%	2.68%
	>70	24,685	24,685	296	1.20%	1.20%	0	0.00%	0.00%	4	0.02%	0.02%	300	1.22%	1.22%
	Total	**130,311**	**130,311**	**1,282**	**0.98%**	**0.98%**	**90**	**0.07%**	**0.07%**	**367**	**0.28%**	**0.28%**	**1,739**	**1.33%**	**1.33%**
9	<26	10,282	10,712	251	2.44%	2.34%	0	0.00%	0.00%	7	0.07%	0.07%	258	2.51%	2.41%
	26-50	39,696	41,353	1,807	4.55%	4.37%	191	0.48%	0.46%	179	0.45%	0.43%	2,177	5.48%	5.26%
	50-70	3,946	4,111	101	2.56%	2.46%	0	0.00%	0.00%	0	0.00%	0.00%	101	2.56%	2.46%
	>70	304	316	6	2.00%	1.92%	6	2.06%	1.97%	0	0.00%	0.00%	12	4.06%	3.89%
	Total	**54,228**	**56,492**	**2,166**	**3.99%**	**3.83%**	**197**	**0.36%**	**0.35%**	**186**	**0.34%**	**0.33%**	**2,548**	**4.70%**	**4.51%**

Table H-4: Estimated Water Quality Improvements Under Option 4																
EPA Region	Baseline Water Quality	Baseline Scenario		Water Quality Improvements by WQI Change												
				0.01 < ΔWQI < 0.1			0.1 < ΔWQI < 0.5			0.5 < ΔWQI			Total Improved Reaches			
		Reach Miles Modeled	Total Reach Miles in RF1 Network	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	Reach Miles	% of Reach Miles Modeled	% of Total Reach Miles	
10	<26	39	40	0	0.00%	0.00%	0	0.00%	0.00%	0	0.00%	0.00%	0	0.00%	0.00%	
	26-50	13,116	13,373	236	1.80%	1.77%	180	1.37%	1.35%	96	0.74%	0.72%	513	3.91%	3.83%	
	50-70	24,190	24,664	1,342	5.55%	5.44%	259	1.07%	1.05%	385	1.59%	1.56%	1,986	8.21%	8.05%	
	>70	30,844	31,448	2,317	7.51%	7.37%	941	3.05%	2.99%	461	1.50%	1.47%	3,719	12.06%	11.82%	
	Total	**68,189**	**69,524**	**3,894**	**5.71%**	**5.60%**	**1,380**	**2.02%**	**1.99%**	**943**	**1.38%**	**1.36%**	**6,217**	**9.12%**	**8.94%**	
Nation	<26	35,137	36,080	932	2.65%	2.58%	13	0.04%	0.04%	132	0.38%	0.37%	1,077	3.06%	2.98%	
	26-50	315,650	325,764	34,773	11.02%	10.67%	5,199	1.65%	1.60%	2,168	0.69%	0.67%	42,140	13.35%	12.94%	
	50-70	182,033	190,537	38,497	21.15%	20.20%	9,735	5.35%	5.11%	3,087	1.70%	1.62%	51,319	28.19%	26.93%	
	>70	94,859	97,662	14,569	15.36%	14.92%	3,548	3.74%	3.63%	1,309	1.38%	1.34%	19,427	20.48%	19.89%	
	Total	**627,679**	**650,043**	**88,772**	**14.14%**	**13.66%**	**18,495**	**2.95%**	**2.85%**	**6,696**	**1.07%**	**1.03%**	**113,963**	**18.16%**	**17.53%**	