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Environmental Impact and Benefits Assessment for Proposed Effluent Guidelines and Standards for the Construction and Development Category

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1 Introduction

The nature of construction activities is to change, often significantly, many elements of the natural environment. Construction activities typically include clearing the land of vegetation, excavating earth, and compacting soil, all of which lead to increased stormwater runoff and higher erosion rates. Construction activities currently affect approximately 590,000 acres in the conterminous United States each year (USEPA, 2008c).

Construction site discharges have been documented to increase the loadings of pollutants to surface waters. The most prominent and widespread pollutant is sediment. The level of sediment is often identified through the measurement of other pollutants in the water body, most notably turbidity, suspended solids, total suspended solids (TSS), suspended sediment concentration (SSC), or settleable solids. Other documented pollutants include metals, nutrients, and polycyclic aromatic hydrocarbons (PAHs). These pollutants can derive from construction equipment and materials or from contamination of a site prior to the start of construction activity. Other possible pollutants include pesticides, other toxic organics, and materials that exert biological oxygen demand (BOD) in surface waters. Construction activities mobilize these pollutants when they disturb soil and increase stormwater runoff within and from a site, making the pollutants available for discharge to surface waters. Pollutants can also enter surface waters during excavation dewatering, construction equipment washout, equipment operation in or near surface waters, and dredging and filling in or near surface waters.

Increases in sediment and other pollutant discharges from construction sites and consequential increases in water body pollutant levels can have adverse impacts on aquatic ecosystems and on human uses of aquatic resources. Ecological impacts from construction pollutant discharges range from temporary to permanent and include both physical impacts on water bodies and biological impacts on aquatic ecosystems. Under a worst-case scenario, a water body may become uninhabitable to some or all varieties of plant and animal life due to high levels of sediment or other pollutants in the water body. Increased sediment and turbidity levels from construction discharges can also impact human uses of water resources, including drinking water supply, recreation, navigation, and fishing, as well as impair proper functioning of stormwater management systems.

EPA reviewed literature pertaining to various aspects of construction site discharges, including their magnitude, impacts on receiving waters, effects on aquatic life, and impacts on human uses of water resources. Information on sediment discharges and their ecological impacts is summarized in *Chapter 2*. *Chapter 3* presents information available on discharges of pollutants other than sediment and on their effects on receiving water bodies.

To quantify national water quality impacts from construction site discharges, EPA used the Spatially Referenced Regressions on Watershed Attributes (SPARROW) modeling methodology. EPA used results from the model to better understand both current conditions and the nature of water quality changes that could result from regulatory actions EPA is proposing to reduce construction site sediment discharges. *Chapter 5* summarizes the regulatory options EPA considered and their associated sediment loading reductions. *Chapter 6*, *Appendix B,* and *Appendix C* discuss the SPARROW methodology and the results of EPA's analysis. Because of data limitations, EPA was not able to conduct a similar analysis for construction site pollutants other than sediment.

The Agency expects that reduced discharges of sediment and other pollutants from construction sites will enhance environmental services provided by the affected water bodies and, as a result, human welfare. Human activities and uses expected to benefit from reduced sediment discharge from construction sites include recreation, commercial fishing, public and private property ownership, navigation, and water supply and use. In analyzing benefits of the regulation, EPA quantified economic benefits of reduced dredging of navigable waterways, reduced dredging of water storage facilities (reservoirs), and reduced drinking water treatment costs. *Chapters 7-9* provide details of the methods used in analyzing these benefit categories and the results of EPA's analysis. Other benefit categories, including reduced flood risk, increases in property values, and decreased wear on industrial water-cooling equipment and commercial fishing, are discussed qualitatively in *Chapter 4*. EPA estimates that Options 1, 2, and 3 will reduce expenditures on navigable waterway dredging by approximately $1, $13, and $27 million per year respectively. Expenditures on reservoir dredging are expected to decrease by $600,000 per year under Option 1, by $17.6 million per year under Option 2, and by $30.6 million per year under Option 3. Reductions in drinking water treatment costs are estimated to amount to $200,000, $7.4 million, and $13.1 million annually under Options 1, 2, and 3, respectively. Overall, EPA expects this regulation to save governments and private entities between $1.8 and $70.9 million each year, depending on the policy option.

EPA also expects that reductions in sediment discharges to surface waters resulting from the regulation will enhance or protect aquatic ecosystems currently under stress. The drop in sediment loading is expected to improve the protection of resident species; enhance the general health of fish, invertebrate, and other aquatic organism populations; increase their propagation to 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, 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. EPA used a meta-analysis of surface water valuation studies to estimate the total value of nonmarket benefits (including use and nonuse values) stemming from the regulation. The Agency estimates that mean total willingness to pay (WTP) for water quality improvements resulting from the regulation ranges from $2 million to $500 million, depending on the regulatory option under consideration. *Chapter 10*, *Appendix D*, and *Appendix E* of this report provide details on EPA's analysis of nonmarket benefits. EPA expects that nonmarket benefits resulting from Option 1 will be approximately $16.6 million per year. Options 2 and 3 have more substantial nonmarket benefits, estimated at $295 million and $400 million per year, respectively. Combining these nonmarket benefits estimates with the avoided cost estimate produces total benefits estimates of $18.4 million per year for Option 1, of $333.3 million per year for Option 2, and of $470.6 million per year for Option 3.

EPA was not able to monetize benefits from reducing discharges of other pollutants that can be present in discharges from construction sites (e.g., nitrogen, phosphorus, PAHs, and metals) due to a lack of quantitative data on the frequency and magnitude of these discharges. *Chapter 3* of this report summarizes the ecological effects associated with discharges of pollutants other than sediment from construction activity. Reducing discharges of these pollutants is expected to generate a wide range of benefits. For example, reducing discharges of nutrients would reduce the rate of eutrophication in receiving waters, leading to fewer beach closings, greater productivity of fisheries in these waters, and greater enjoyment of these water resources. Reducing discharges of toxic pollutants such as PAHs and metals from construction sites would reduce the level of these pollutants in the affected water bodies, leading to healthier aquatic communities and lower pollutant concentrations in water body sediment. Because sediment dredged from water body beds or sludge waste from drinking water treatment is more

costly to dispose of when it contains elevated levels of toxic pollutants, lower levels of toxic pollutants are expected to lower the costs of sludge and dredged materials disposal. As these pollutants also factor into the determination of the water quality of a surface water body, reducing ambient concentrations of these pollutants is likely to increase the value of affected water resources.

2 Sediment

Sediment is the most commonly documented pollutant in construction site discharges. Dramatic increases in sediment loadings downstream of construction sites have been documented by dozens of studies over the past four decades. An early and frequently cited study by Wolman and Schick (1967) performed in the Baltimore, Maryland area found that suspended sediment concentrations in streams receiving construction site discharges were anywhere between 2 and 200 times greater than those in streams in rural or wooded areas.

Sediment from construction sites can consist of both inorganic and organic materials. Sediment influences several measures of water quality and pollutant levels:

- Total suspended solids (TSS) – a measure of the amount of suspended inorganic and organic material in the water column, usually expressed in milligrams per liter.
- Turbidity – a measure of the ability of light to penetrate water. Turbidity is typically measured with a nephelometer (a type of turbidimeter) and expressed in nephelometric turbidity units (NTUs). Turbidity levels are related to the quantity, size, shape, and color of particles suspended in the water column (many small particles reduce light penetration more than a few large particles of the same mass). Higher NTU levels mean that less light is able to penetrate water.
- Total dissolved solids (TDS) – a measure of the dissolved particles in water, usually expressed in milligrams per liter. It is often used as an indicator of drinking water quality.

Sediment can also settle out of the water column, creating a layer of sediment on a water body's bed. This process is commonly called siltation or sedimentation, and the sediment can be referred to as settleable solids.

Episodic precipitation events are the primary cause of construction site sediment discharges to surface waters. Most sediment discharge, therefore, takes place during or shortly after precipitation events. Once a precipitation event ceases, discharges from construction sites generally cease within a relatively short time period. Individual construction sites are transient as well, with activity typically lasting several months to several years. Sediment discharges due to active construction at an individual site are therefore typically transient, too. However, impacts from a construction site's sediment discharges may persist beyond the lifespan of a precipitation event or of a site's active construction. Sediment discharges typically contain large fractions of persistent matter that can remain in surface waters for long periods of time. During this time, sediments can shift multiple times between suspension in the water column and deposition on a water body bed, depending on currents and other disturbance factors. As time passes, the sediments can migrate downstream and impact new waterbodies far from their original point of entry into the surface water network.

Because of sediment's ability to migrate downstream, a waterbody 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 discharges, even though the timing of individual precipitation events and locations of individual construction sites within the watershed may change from year to year.

Some water bodies contain aquatic ecosystems that are adapted to sediment influx, whereas others contain ecosystems adapted to low levels of sediment input. The appropriate level of sediment varies from

waterbody to waterbody. However, even ecosystems adapted to sediment input may become impaired if sediment input is excessive or if the ecosystem's resiliency is taxed by other stressors.

High levels of sediment can have profound effects on aquatic ecosystems, impairing photosynthetic activity, reducing available food, and burying habitat, thereby causing fish and other organisms to relocate or die. The sections below provide additional detail on construction site sediment discharges and the consequences of increased sediment loads to receiving waters and their ecosystems. For a detailed discussion of the process of sediment discharge from construction sites, see EPA's *Development Document for Proposed Effluent Guidelines and Standards for the Construction and Development Category* (USEPA 2008b). Elevated sediment loadings can create many adverse economic impacts, including decreased reservoir capacity, increased navigable channel dredging, increased drinking water treatment costs, and decreased water body recreational value. These and other economic impacts are discussed in *Chapter 4* and *Chapters 7-10*.

2.1 Sediment Behavior in Surface Waters

Sediments, once eroded from land surfaces and delivered to surface water bodies, are transported and ultimately deposited within the riverine flow system (including banks, floodplains, or terraces); within lakes, reservoirs, or other impoundments; or in estuaries, bays, or elsewhere on the continental margins. Factors influencing sediment impact duration are the intensity, quantity and composition of the sediment discharges and the nature of the receiving water. Some surface waters are able to purge or absorb sediment inputs within relatively short time frames. Other surface waters allow sediment accumulations to persist for long periods of time.

2.1.1 Important Sediment Characteristics

Important characteristics of sediments that influence transport and fate include the size, shape, density, fall velocity, and concentration of sediment particles (Shen and Julien 1993). Commonly used definitions of particle size include *nominal diameter*, or diameter of a spherical particle of equivalent volume, and *sieve diameter*, which is the smallest square sieve opening through which the particle will pass (units of mm or μm). *Table 2-1* presents the broad categories of the sediment size scale.

Table 2-1: 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)

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

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

Where:

ω = settling velocity (cm s^{-1})

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

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

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

ρ_w = density of water (g cm^{-3})

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

d_s = particle diameter (cm)

This formula indicates 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 particle size distribution describes the percentage by weight of sediment materials in each size category for a given sediment sample. The sediment size d_{50}, or median grain size, is the particle size for which 50 percent of particles by weight are larger and 50 percent are smaller, and is often used to summarize a sediment sample. Similarly, d_{90} indicates a relatively coarse fraction, for which 90 percent of particles are smaller.

2.1.2 Sediment Transport

Sediment in Streams and Rivers

The transport and fate of sediments within river and stream channels are controlled by two general sets of influences. These are the factors determining the stream transport capacity and the rate of supply of the sediments themselves, along with their physical and chemical characteristics.

Sediments conveyed by stream channels, known as *total sediment load*, are typically partitioned into *bed load* and *suspended load*, although the boundary separating these regimes is not rigid, but rather determined by flow conditions. Sediment particle size is the primary distinguishing factor, with larger particles moving as bed load and finer particles as suspended load.

Bed load consists of the movement of particles along the streambed, by rolling, sliding, and hopping. Hopping, or saltation, involves the brief suspension of sediment particles by turbulent flow. Particles moving as bed load typically consist of sand and coarser size fractions, and the bed layer thickness is approximately twice the bed load particle diameter. Bed load is, in most settings, intermittent, and bed load particles may not move for long periods during low flows or between flood events.

Suspended sediment load is “the rate of movement of sediment particles that are supported by the turbulent motion of the stream flow” (Shen and Julien 1993) and consists primarily of the clay and silt fractions of the particle size distribution. It is commonly measured as Suspended Sediment Concentration (SSC) or, alternatively, as Total Suspended Sediment (TSS). 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) has 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. The *total sediment load* consists of the sum of bed load and suspended load, respectively. The total sediment flux (total mass of sediment passing through a given stream cross-section per unit of time) is typically difficult to measure directly. To obtain estimates of total sediment flux, bed load and suspended load are sampled, and the total flux is estimated by multiplying each sampled concentration by the respective volume of flow.

Several hydraulic and geomorphologic factors determine the capacity of a given stream reach to mobilize and to transport sediment, both as bed load and suspended load. A partial list of these factors includes 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 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:

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

Where:

q_s = sediment transport per unit stream width (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)*, with values of c_3 ranging from 3.3 to 3.95. Many, if not most, of the hydraulic factors controlling sediment transport capacity, including channel geometry, depth and velocity of flow, vary extensively in time and space within a given river system.

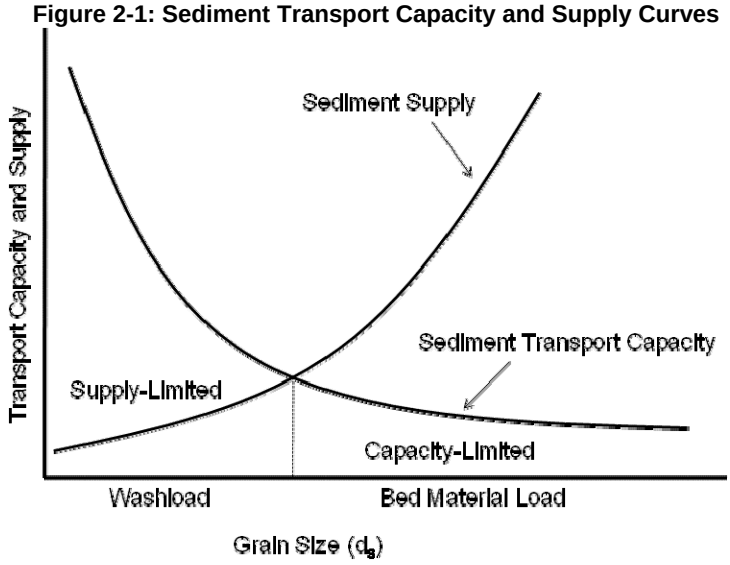
Most North American rivers and streams possess a strong seasonal discharge cycle, with spring discharge volumes typically many times larger than late summer – autumn flows. Intense or prolonged rainfall events can also generate flood pulses of hourly to daily duration, which often have significant turbulent energy. In addition, as Eq. 1-2 indicates, sediment transport capacity is a non-linear function of flow-related variables. The movement of sediments, both as bed load and as suspended load, is thus highly non-uniform in time for most river systems. The majority of annual sediment flux, and particularly the movement of coarse or highly cohesive sediment particles may occur over a relatively short period of time within a single flood event. Between such events, sediments are typically stored within the channel. In developing annual or long-term sediment budgets for specific river reaches, it is therefore necessary to utilize sediment concentration samples and associated flow volumes representative of the entire range of annual flow conditions. Long-term annual budgets are derived by estimating the sediment mass fluxes for

each rate of flow, and weighting these fluxes by the fraction of the year for which the flow conditions typically occur.

Erosion and deposition of sediments within a river course also exhibit spatial patterns strongly related to stream morphology. River reaches having smaller cross-sectional flow area, steeper slopes and higher flow velocities, typical of upper catchments, discourage the deposition of sediments. By contrast, wider channels with lower bed slopes and associated velocities, typical of downstream reaches, act as regions of relative sediment deposition. Sediments are deposited preferentially in regions of low-energy flow, including pools and the inside of bends, where bars form (Chapra 1977). Such deposition zones are not permanent features of the watercourse, since these deposits can be scoured and moved downstream during major flow events.

Hydraulic and geomorphologic variables provide one set of controls on the sediment transport capacity of a given river reach. Actual sediment transport can also be regulated by the rate of supply of sediments (*Figure 2-1*, Julien 1995). Within a particular reach, observed sediment fluxes can originate either from land surface (erosion) processes adjacent to the reach, from streambank erosion, from upstream reaches, or from the re-mobilization of sediments previously deposited within the reach.

Figure 2-1: Sediment Transport Capacity and Supply Curves



Source: Julien (1995)

One consequence of the competing controls on sediment transport is that increases/reductions in land surface erosion and sediment delivery rates may not result in directly proportional increases/decreases in observed sediment loads in downstream reaches. For example, a reduction in sediments delivered to a particular reach resulting from improved land management may result in decreased total sediment loads within that reach, resulting in turn in less stream energy expended on turbulent suspension of sediment. This energy is then available for the re-suspension of sediments previously deposited within the reach, which will be reflected in sediment concentrations at points downstream. In situations where sediment supplies are strongly limited, such as in reaches below dams or other sediment-trapping impoundments, flows may possess considerable excess transport capacity. Such *sediment-starved* flows may entrain gravel or other large particles, and result in bank erosion and bed incision (Kondolf, 1995).

The episodic movement of sediments within river systems at seasonal, annual, decadal and even longer timesteps must be taken into account in efforts to link land use and management policies with in-stream water quality indices. The geomorphologic perspective on erosion and sedimentation emphasizes an understanding of the long-term frequency–magnitude relationships of the flow events that are largely responsible for re-distributing sediments within river systems, thereby *resulting* in observed hydraulic and topographic factors. *Effective discharge* (Wolman and Miller, 1960) is defined as the rate of discharge associated with the greatest contribution to sediment transport over time (alternatively, the flow instrumental in determining channel morphology). This is often found to be the maximum product of discharge magnitude and event frequency. For temperate regions within the U.S., effective discharge is often found to correspond to flow events having a recurrence interval of roughly 1.5 years (Dunne and Leopold, 1978).

Sediment in Lakes and Reservoirs

Lakes and reservoirs have lower flow velocities and greater depth of flow than streams and rivers and often act as deposition locations (sinks) for sediments. The lake and reservoir flow environment more closely approximates the still-water conditions under which Stokes' law (*Section 2.1.1*) 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 water body. Several factors influence settling and transit times, including the origin of the impoundment (natural versus artificial), size, shape, depth, and regulation of outflow (Chapra, 1977). These characteristics vary widely among lakes and reservoirs.

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). Current and wind-induced turbulence 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).

Sediment in Estuaries

Estuaries, like lakes and other impoundments, exhibit wide variation in benthic geometry, residence times or flushing rates, vertical mixing and stratification, wave exposure, and other factors that govern the transport and fate of sediments. Within estuary systems, flow regimes transition first from unidirectional turbulent channel flow controlled primarily by topographic gradient and discharge rate to a tidal river reach in which downstream flow is influenced by the tidal cycle. The flow regime transitions again in the estuary proper, in which discharge and tidal forces are offsetting and then again in the bay or open ocean where tidal, wave, or combinations of these forces dominate. As sediment-laden freshwater decelerates and encounters tidal cycles upon entering 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 estuaries, lakes and other impoundments, deposited sediments can be disturbed and re-distributed due either to extreme natural events, including floods and high winds, or by human impacts and interventions.

2.1.3 Turbidity

Turbidity is a non-conventional pollutant and an important dimension of water quality in streams, lakes and estuaries that is closely related to, but distinct from Total Suspended Solids (TSS). Turbidity refers generally to the extent to which light can penetrate the water column, with greater depths of light transmission associated with lower turbidity and higher water quality. Turbidity is commonly measured using a Secchi disk, lowered into the water column, and the depth at which the black-and-white pattern on

the disk remains visible is the Secchi depth. It is typically assumed that light can penetrate to roughly 2-3 times the Secchi depth. Turbidity is also measured in nephelometric turbidity units, using a turbidimeter containing a photocell.

Turbidity is often used as an indicator of photosynthetic activity, although all types of particulate matter within the water column will affect turbidity measurements. Particulate matter consists of mineral particles ("sediment"), algae and related photosynthetic organisms, and organic detritus. Within most lakes, photosynthetic organisms are the major source of turbidity, although sediments can be an important factor in determining water clarity. Sediments can be introduced via erosion and riverine transport processes, particularly during floods; by human activities (e.g., dredging), by the actions of bottom-feeding fish such as carp; or by re-suspension due to wind and wave action. In most water bodies, turbidity also has a seasonal component, as phytoplankton concentrations vary due to water temperature, dissolved oxygen and the availability of nutrients.

2.2 Documented Relationship between Construction and Sediment

There are a significant number of studies addressing sediment discharges from construction sites, as well as multiple state water quality reports that document construction sites associated with surface water impairment with sediment. EPA conducted a literature search that yielded 58 studies on this topic: 17 case studies that measure sediment increases due to construction, 5 case studies that measure this and also study the impacts on the aquatic ecosystem, and 36 studies that qualitatively discuss cases of construction sites increasing sediment levels in surface waters and impacting aquatic life. These studies are summarized in the sections below.

2.2.1 Impacts of Construction Site Discharge on Sediment Levels in Surface Waters

Table 2-2 lists 17 studies which focus on quantifying impacts of construction site discharges on sediment levels in surface waters. EPA found an additional 38 documents that discuss this topic qualitatively. These documents are discussed at the end of this section.

The earliest of the quantitative studies EPA located was conducted from 1961 to 1964 by the U.S. Geological Survey (Vice et al. 1969). Study of this topic has continued to the present with two studies published in 2006 (Cleveland and Fashokun and Hedrick et al.). Overall, this body of studies finds that construction activity increases sediment discharge, subsequently increasing the amount of sediment in affected water bodies. An exception is Hedrick et al. (2006) which finds no statistical difference in sediment levels in a stream affected by construction compared to those of an unaffected stream. The authors attribute this observation partially to the successful implementation of sediment abatement practices at the construction site and partially to the fact that construction took place during the winter when the ground was mostly frozen (Hedrick et al. 2006) and therefore more resistant to erosion.

Table 2-2: Studies Quantifying Sediment Increases in Surface Waters

Author(s)	Year	Study Name
Barrett et al.	1995	Effects of Highway Construction and Operation on Water Quality and Quantity in an Ephemeral Stream in the Austin, Texas Area
Carline et al.	2003	Effects of Bridge and Road Construction on Spring Creek: Route 26 Transportation Improvements Project
Cleveland and Fashokun	2006	Construction-Associated Solids Loads with a Temporary Sediment Control BMP
Daniel et al.	1979	Sediment and Nutrient Yield from Residential Construction Sites

Table 2-2: Studies Quantifying Sediment Increases in Surface Waters

Author(s)	Year	Study Name
Duck	1984	The Effect of Road Construction on Sediment Deposition In Loch Earn, Scotland
Hedrick et al.	2006	Impact of Tunnel Reconstruction on Stream Water Quality in Great Smoky Mountains National Park
Helsel	1985	Contributions of Suspended Sediment from Highway Construction and Other Land Uses to the Olentangy River, Columbus, Ohio
Lubliner and Golding	2005	Stormwater Quality Survey of Western Washington Construction Sites, 2003-2005
Nelson and Booth	2002	Sediment Sources in an Urbanizing, Mixed Land-Use Watershed
Novotny and Chesters	1989	Delivery of Sediment and Pollutants from Nonpoint Sources: A Water Quality Perspective
Owens et al.	2000	Soil Erosion from Two Small Construction Sites, Dane County, Wisconsin
Reed	1980	Suspended-Sediment Discharge in Five Streams Near Harrisburg, Pennsylvania, Before, During, and After Highway Construction
Stephens et al.	1996	Potential Effects of Coal Mining and Road Construction on the Water Quality of Scofield Reservoir and Its Drainage Area, Central Utah, October 1982 to October 1984
Vice et al.	1969	Sediment Movement in an Area of Suburban Highway Construction, Scott Run Basin, Fairfax County, Virginia, 1961-64
Ward and Appel	1988	Suspended-Sediment Yields in the Coal River and Trace Fork Basins, West Virginia, 1975-84
Wong	2005	Water Quality in the Haiawa, Haiku, and Kaneohe Drainage Basins Before, During, and After H-3 Highway Construction, Oahu, Hawaii, 1983-99
Yorke and Herb	1978	Effects of Urbanization on Streamflow and Sediment Transport in Maryland

Table 2-3, *Table 2-4*, and *Table 2-5* summarize findings from the studies. Depending on the objective of the study, sediment in receiving waters is measured either as a concentration (*Table 2-3*) or as a total yield from the surface water's watershed basin (*Table 2-4*). Sediment concentrations are measured in milligrams per liter (mg/L) and units of mass are used to measure sediment yield. 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. Several studies also measure turbidity in the receiving water (*Table 2-5*), as it is closely related to suspended sediment levels and is also an indicator of water quality. Turbidity, a measure of the extent to which light is scattered, is commonly measured in NTUs, but also may be measured in other units such as formazin turbidity units (FTUs). Both units measure the light-scattering properties of the water, but each is specific to the measurement device and its calibration. In many of the studies, measurements are taken from the surface water area adjacent to the construction site, though in some cases measurements are taken further downstream.

Table 2-3: Summary of Sediment Concentration Findings from Selected Studies

Study	Location	Sediment Concentration (mg/L)	
		Unaffected Site	Affected Site
Barrett et al. (1995)	Austin, TX	34.8	179.0
Barton (1977)	Ontario, Canada	2.0	50.0
Chisholm and Downs (1978)	Boone County, WV	—	3,051.2
Cleveland and Fashokun (2006)	Harris County, TX	122.5	57.3
Cleveland and Fashokun (2006)	Harris County, TX	1,227.1	1,410.3
Cline et al. (1982)	Rocky Mountains, CO	16.4	10.8
Cline et al. (1982)	Rocky Mountains, CO	2.2	18.9
Cline et al. (1982)	Rocky Mountains, CO	3.8	91.0
Cline et al. (1982)	Rocky Mountains, CO	2.6	121.6
Duck (1984)	Loch Earn, Scotland	1.4	134.3
Helsel (1985)	Olentangy River, OH	125.0	261.0
Line and White (2007)	Piedmont Region, NC	349	6,170
Lubliner and Golding (2005)	Northwest WA	—	33.7
Lubliner and Golding (2005)	Northwest WA	—	131.2
Lubliner and Golding (2005)	Northwest WA	—	628.7
Owens et al. (2000)	Dane County, WI	—	2,400.0
Owens et al. (2000)	Dane County, WI	—	15,000.0
Wong (2005)	Honolulu, HI	—	7.0
Young and Mackie (1991)	Northwest Territories, Canada	15.0	93.0

Table 2-4: Summary of Sediment Yield Findings from Selected Studies

Study	Location	Sediment Yield		Units
		Unaffected Site	Affected Site	
Carline et al. (2003)	Centre County, PA	—	182.0	metric tons/year
Daniel et al. (1979)	Southeast WI	—	19.2	metric tons/hectare
Downs and Appel (1986)	Coal River, WV	107.6	220.0	tons/sq. mi.
Nelson and Booth (2002)	Northwest WA	—	43	tons
Reed (1980)	Harrisburg, PA	71.0	294.3	tons
Selbig et al. (2004)	Dane County, WI	42.2	60.3	tons
Selbig et al. (2004)	Dane County, WI	3.8	4.2	tons
Stephens et al. (1996)	Scofield, UT	0.2	0.7	metric tons/sq. km.
Vice et al. (1969)	Fairfax, VA	2,890.0	33,320.0	tons
Ward and Appel (1988)	Coal River, WV	746.0	1,480.0	tons/sq. mi.
Wolman and Schick (1967)	Maryland	—	97,333.3	tons/sq. mi./year
Yorke and Herb (1978)	Rock Creek, MD	—	32.7	tons/acre

Table 2-5: Summary of Turbidity Findings from Selected Studies

Study Author	Location	Turbidity (NTU)	
		Unaffected site	Affected site
Barrett, et al. (1995)	Austin, TX	28	112
Chisholm & Downs (1978)	Boone County, WV	2.3[1]	7.1[1]
Cleveland & Fashokun (2006)	Harris County, TX	95.6[2]	71.9[2]
Cleveland & Fashokun (2006)	Harris County, TX	141.1[2]	159.2[2]
Lubliner & Golding (2005)	Northwest WA	2.2	2.2
Lubliner & Golding (2005)	Northwest WA	5.2	6.4
Lubliner & Golding (2005)	Northwest WA	5.5	5.2
Lubliner & Golding (2005)	Northwest WA	6.4	25.7
Lubliner & Golding (2005)	Northwest WA	9.8	45.0
Lubliner & Golding (2005)	Northwest WA	17.6	12.0
Wong (2004)	Honolulu, HI	4.0[3]	4.9

1 Averages of multiple measurements at comparable measurement sites on two tributaries to the Little Coal River.
2 Measured in FTU.
3 "Unaffected site" measurement is measurement before start of construction activity.

Because construction site sediment loadings are highly dependent on site conditions, rainfall, nature of construction activity, presence and effectiveness of sediment abatement practices, and many other factors, the observations summarized here are only intended to give a sense of the range of documented sediment loadings. The enormous variation in measurements from different sites and different studies underscores the difficulty of drawing general conclusions about sediment loadings from all construction sites based on information available in the literature. EPA estimates more than 590,000 acres of land, encompassing a wide variety of sediment discharge conditions, undergo construction activity each year in the United States (USEPA 2008c). The studies summarized here provide valuable information on actual construction sites, but do not describe conditions resulting from all current construction activity taking place in the United States. Therefore, EPA did not rely on this collection of studies to quantitatively estimate current national water quality impacts from construction site sediment discharges and how they would decrease under the proposed regulatory options. Instead, EPA used the Spatially Referenced Regressions on Watershed Attributes (SPARROW) water quality modeling methodology to estimate surface water sediment levels associated with construction activity. This methodology and the results of the analysis are discussed in more detail in *Chapter 6* of this document.

However, it is possible to conclude from the available studies that construction activities do lead to higher levels of sediment discharge to surface waters and higher levels of sediment in surface waters. All studies comparing areas under construction to areas unaffected by construction conclude that construction areas produce higher sediment loadings than their undisturbed counterparts. Although certain observations in some studies, notably Cleveland and Fashokun (2006) and Cline et al. (1982), show some undisturbed areas producing higher sediment levels than construction-affected sites, these particular observations are associated with special circumstances noted by the authors. In Cleveland and Fashokun (2006), TSS levels in a drainage ditch upstream of a construction site were higher than levels downstream of the site only during nonstorm periods, whereas most sediment leaves construction sites during or shortly after precipitation events. The authors note that, based on their observations, construction activity caused a six-fold increase in total sediment leaving a construction site. In Cline et al. (1982), a reservoir between the observation points upstream and downstream of a construction site released clear water into the affected stream, diluting TSS concentrations at the downstream observation point.

In total, 15 of the 17 studies measuring sediment levels in surface waters near or downstream of construction sites find that construction activity contributed significantly to downstream sediment levels.

Of the two studies that do not document a consistent impact, Lubliner and Golding (2005) find that of 183 construction sites visited, only 44 were discharging at the time of their visit. The authors attribute the absence of discharge at 139 sites to lower intensity rainfall, permeable soils, the use of sediment abatement practices, and the difficulty of timing their visits to coincide with discharge events at the sites they had chosen. Of the 44 discharging sites, 6 were discharging directly to surface waters, 2 of which caused noticeable increases in TSS and turbidity levels downstream of the construction site. The remaining 38 sites allowed stormwater to seep into the ground or discharge to a city stormwater system (Lubliner and Golding 2005). The second study, by Hedrick et al. (2006), finds no statistical difference between sediment loadings at affected and unaffected sites, which they attribute to the effective implementation of sediment abatement practices and the resistance of frozen ground to erosion.

An early study by Vice et al. (1969) finds that highway construction areas constituted between 1 and 10 percent of a total watershed's basin area at any given time but contributed 85 percent of that watershed's total sediment yield. Measurements were taken nearly 2 miles downstream from the construction area, indicating that the construction impacts were not localized to the active construction site, and discharged sediment was carried downstream. Reed (1980) monitored a highway construction site in central Pennsylvania before, during, and after construction activity. He finds that sediment discharge increased two- to four-fold during the active construction period over pre-construction levels, even in the presence of sediment abatement practices, but quickly returned to preconstruction levels once construction activity was complete. Daniel et al. (1979) investigated water quality impacts of residential construction sites in southeastern Wisconsin and find that construction activities yielded significantly higher amounts of both sediments and nutrients (see the discussion of nutrients in *Chapter*) than comparable agricultural land use. This is notable because agricultural land use has been identified as an important source of water quality degradation through discharges of both sediments and nutrients, as well as other pollutants (Clark et al. 1985), though it encompasses a larger percentage of total land area in the United States on an annual basis than construction activity. Nelson and Booth (2002) calculate in their study of Northwest Washington that construction sites had the highest sediment yield per unit area of all land uses they examined.

More recently, Barrett et al. (1995) measured water quality indicators upstream and downstream of a highway construction site in the Austin, Texas area. They find that sediment yields from the site increased due to construction activity and that turbidity levels in the impacted stream increased more than TSS levels. The same study also finds increases in metal contamination downstream of construction sites that were closely correlated to sediment discharge levels (see the discussion of metals in *Chapter 3).* Similar to Reed (1980), Barrett et al. (1995) finds these increases to be temporary, dissipating quickly after construction ended.

Another recent study by Owens et al. (2000) examining water quality impacts from one residential and one commercial construction site in Wisconsin reports substantial increases in TSS concentrations during the construction phase at both sites. The commercial site created higher sediment concentrations than the residential site, though the authors believe this observation is partially explained by the fact that the residential construction took place during the winter, when rainfall intensity tends to be much lower, whereas the commercial construction took place during the summer, when rainfall intensity is higher. A 2006 study by Cleveland and Fashokun on construction of a highway in Texas finds that construction activity resulted in a six-fold increase in total sediment leaving the construction site, despite the presence of a rock filter dam constructed to reduce sediment discharge from the site. Most recently, Line and White (2007) compared water quality in a developing area with an analogous water in an undeveloped area. Their findings suggest that construction activities significantly alter sediment loadings above increases

that may be attributed to overall changes in land use, particularly during the initial land-clearing phase. Overall TSS export from the developing area was 95 percent greater than from the undeveloped area.

Five of these case studies measure turbidity as well as TSS levels in surface waters affected by construction site sediment discharges. Barrett et al. (1995) recorded a substantial increase in turbidity from 28 to 112 NTU at the affected measurement site, noting that this is greater than the increase in TSS due to the ineffectiveness of construction site silt fences for reducing turbidity. Chisholm and Downs (1978) note higher turbidity in a reach that was receiving construction sediment discharges compared to a nearby reach that was not. Similar to their TSS findings, Cleveland and Fashokun (2006) find that turbidity was higher in the upstream site during non-storm conditions but during storms was higher at the downstream site. Downstream turbidity values during construction activity were also higher compared to measurements taken before construction started. As mentioned earlier, Lubliner and Golding (2005) note an increase in sediment levels at two of six construction sites discharging to surface waters which led to four- to five-fold increases in turbidity levels. Wong (2004) measured turbidity before and during highway construction activity in Hawaii and finds that average turbidity was higher during the construction period over a range of different sites and time periods.

In addition to measuring increases in surface water TSS concentrations and turbidity levels due to construction site discharges, seven of the above studies discuss increases in sediment deposition, or sedimentation, also resulting from increases in sediment discharges to surface waters. Duck (1985) recorded 1,824 metric tons of sediment deposited on a lake bed during a two month construction period. The author estimates that the lake, under natural conditions, would have otherwise taken 20 to 25 years to accumulate this quantity of sediment. Similarly, Stephens et al. (1996) note increased sediment deposition in the Scofield Reservoir in Utah due to construction activities discharging sediment to its tributaries.

As well as increasing the quantity of sediment on the surface water bed, sedimentation can also change the composition of a surface water bed's substrate. Vice et al. (1969) estimate that 50 percent of the sediment discharged from observed construction activity was deposited in receiving surface waters. Barton (1977) finds that deposition increased 10-fold due to construction site discharges to a small stream in Ontario, resulting in decreased proportions of organic matter in the sediment. Chisholm and Downs (1978) recorded 8 to 10 inches of silt and clay deposited on the bed of a small stream in West Virginia. Cline et al. (1982) recorded an increased percentage of fine sediments in the substrate of the observed surface water.

None of these studies evaluate how long sediment deposits remain on surface water beds. Barrett et al. (1995) note that increased sediment deposition in the creek under study was ameliorated within one year of cessation of construction activity. Sediment deposited in surface waters with higher energy currents such as those found in many rivers and streams is more likely to be re-suspended in the water column and carried downstream than sediment deposited in water bodies with lower energy currents, such as many lakes and reservoirs. These water bodies can lose capacity over time due to sediment accumulation.

In addition to the studies presented above, there are a significant number of other cases in which construction activity has been implicated in elevated downstream sediment loadings and ecological degradation. A number of these cases have been reported by state environmental protection departments as part of their assessment of their state's surface water quality. A review of the state of Ohio's water quality reports from the early 1990s through 2006 yielded 31 reports specifically 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). Ohio EPA (1997f), a 1995 water quality study

of the Twin Creek watershed, focuses on construction as the source of pollution in the Twin Creek basin. A golf course construction site had several negative impacts on water quality and aquatic life in several of the water bodies evaluated. Turbidity in the water column was observed up to 10 miles downstream from the construction site and was attributed to uncontrolled construction site stormwater discharges to the surface water. The authors caution that construction site discharges pose a threat to this otherwise high-quality watershed, because though each construction site is short-lived, ongoing development can make construction's influence on water quality pervasive and substantial over time.

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

In addition to these state reports, the World Wildlife Fund (WWF) identifies construction sediment discharges as a threat to the health of the Tennessee, Cumberland, and Mobile river basins (Buckner et al. 2002).

Helsel (1985) and Ward and Appel (1988) both note that although construction site sediment discharges increase sediment levels in nearby water bodies, the small area of construction sites relative to the larger area of the entire watershed render these increases less perceptible at a larger scale.

The studies summarized in this section provide strong evidence linking the land disturbance associated with construction activities to elevated sediment and turbidity levels in downstream surface waters. These increases have been documented over the past half-century in many regions of the country, over a wide range of site conditions, in all seasons, and with varying levels of sediment abatement practices, underscoring the widespread nature of the problem of sediment pollution from construction sites.

2.2.2 Impacts of Construction Site Sediment Discharge on the Aquatic Ecosystem

Sediment has multiple well-documented impacts on aquatic ecosystems. These impacts are described in more detail in *Section 2.2.* EPA's literature search yielded six studies that measure the impact of construction sediment discharges on both water quality and the aquatic ecosystem. *Table 2-6* lists these studies.

Table 2-6: Studies Quantifying Sediment Increases and Their Impacts on Aquatic Life

Author(s)	Year	Study Name
Barton	1977	Short-term Effects of Highway Construction on the Limnology of a Small Stream in Southern Ontario
Chisholm and Downs	1978	Stress and Recovery of Aquatic Organisms as Related to Highway Construction Along Turtle Creek, Boon County, West Virginia
Cline et al.	1982	The Influence of Highway Construction on the Macroinvertebrates and Epilithic Algae of a High Mountain Stream
Selbig et al. (USGS)	2004	Hydrologic, Ecologic, and Geomorphic Responses of Brewery Creek to Construction of a Residential Subdivision, Dane County, Wisconsin, 1999–2002
Wolman and Schick	1967	Effects of Construction on Fluvial Sediment, Urban and Suburban Areas of Maryland
Hunt and Grow	2001	Impacts of Sedimentation to a Stream by a Non-Compliant Construction Site

Like the studies described in *Section 2.2.1,* which focus primarily on the magnitude of sediment yield or surface water sediment concentration change created by construction sites, these six studies provide

quantitative measurements of sediment loadings to impacted surface waters as part of their assessment of the biological communities in the affected water bodies. The magnitudes of the sediment loadings documented by these studies are listed in *Table 2-3*, *Table 2-4*, and *Table 2-5*. All of the studies find that sediment yields or concentrations in surface waters affected by construction sites were higher than those in unaffected waters.

Wolman and Schick (1967, p. 451) find that increased sediment deposition led to changes in the structure of the aquatic community immediately downstream of construction activity. They list "…blanketing of bottom-dwelling flora and fauna, alteration of the flora and fauna due to changes in light transmission and abrasive effects of sediment, and alterations of species of fish due to changes produced in the flora and fauna upon which the fish depend…" as the impacts observed in a stream reach downstream of construction activity.

Barton (1977) and Chisholm and Downs (1978) conducted studies on the effects of highway construction on aquatic ecosystems in southern Ontario and West Virginia, respectively. Barton (1977) notes that standing fish crops were significantly reduced immediately downstream of construction activity that caused TSS concentrations to increase from 2 mg/L to 50 mg/L. The study also notes that the composition of the benthic macroinvertebrate community shifted, though no species were eliminated. Chisholm and Downs (1978) find more severe impacts to the benthic macroinvertebrate community, noting a reduction in total abundance of organisms and the elimination of certain benthic species due to TSS concentrations greater than 3,000 mg/L caused by construction site discharges.

Cline et al. (1982) evaluated the effects of highway construction on the macroinvertebrate community in a Rocky Mountain stream and found it to be altered by construction, though not as severely as anticipated. Sediment concentrations downstream of areas affected by construction activity were 9 to 45 times greater than concentrations downstream of unaffected areas. Impacts observed were similar to those found by Barton (1977) and Chisholm and Downs (1978) and included a reduction in density, abundance, and diversity, though no elimination of species was found. The authors also note reduced organic content in the epilithon (rock-coating material) and fewer algal species.

In three of these studies, the biological effects of increased sediment levels adjacent to or a short distance downstream of the construction site are found to be temporary, typically lasting less than one year after the cessation of construction activity (Barton 1977; Chisholm and Downs 1978; Cline et al. 1982).

Selbig et al. (2004) find no statistical difference between the sediment loadings from a site affected by construction and one unaffected by construction, though the construction-affected site did have marginally higher mean sediment levels measured from 30 to 100 feet downstream of construction activity. They also find no substantive changes to the biological communities in the affected stream reaches. The authors attribute this lack of observable impact to the effective implementation of several different sediment abatement practices at the construction site.

In a 1999 study of the water impact of noncompliant construction activities on Ohio's Turkey Creek, Hunt and Grow (2001) attributed the significant decrease in fish species diversity, quantity, and weight to the significant increase of silt and clay in the creek's substrate due to uncontrolled discharge from the construction site. The number of fish species decreased from 26 at the upstream site to 19 at the downstream site. The number of fish in the equally sized study areas decreased more drastically, from 525 to 230. Pollutant-tolerant species increased by 237 percent, while pollutant-intolerant species decreased by 33 percent. The weights of fish also dropped by 62 percent. The overall rating of the aquatic habitat declined significantly at the downstream site due to these negative impacts attributed to uncontrolled construction site discharge.

In addition to these studies that examine the effects of quantified increases in sediment levels on aquatic ecosystems, there are multiple studies that recognize upstream construction activity as the source of downstream aquatic life degradation in specific locations. Water quality reports from the Ohio EPA evaluate biological indicators of water quality and, where possible, identify the water body's sources of impairment. Multiple reports identify fish and macroinvertebrate communities impaired by elevated suspended or deposited sediment levels deriving from construction activity upstream. Several types of construction are identified as sediment 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). In particular, Ohio EPA (1997f), a 1995 water quality study of the Twin Creek watershed, finds that construction of a golf course severely impacted nearby stream habitat, where the investigators found none of the live fingernail clams that were present throughout the rest of the watershed but found instead pollutant-tolerant midges which were not common in the overall watershed. They attribute this to the four to six inches of silt that had been introduced to the stream bed from construction discharges.

The state of California has also expressed concern about discharges from construction sites and specifically discusses 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 World Wildlife Fund documents the most severe aquatic ecosystem impact, describing an incident in Tennessee where road construction crews failed to implement appropriate sediment controls and a rain event deposited more than four feet of sediment on the bed of a nearby creek, eliminating aquatic organisms and contaminating the water supply of a downstream town (Buckner et al. 2002).

The studies summarized in *Sections 2.2.1* and *2.2.2* above document impacts associated with known construction sites. Given the large number of construction sites active every year, the large number of sediment-impaired waters in the United States (see *Section 2.3.1*), and limitations in resources available for impact study and documentation, the documentation of impacts summarized here cannot describe all impacts associated with sediment discharge from construction sites or all known incidences of impact. This fact is made clearer by reports of sediment discharge from construction sites from members of the public that have not been published as studies. For example, several reports were submitted as sworn testimony in *(NRDC et al. v. U.S. EPA*, 2005a, 2005b, 2005c, 2005d, 2005e). These reports describe various testifier-observed construction site impacts including sediment-laden and turbid discharges, elevated surface water turbidity and sediment levels, creek and lake sedimentation, loss of recreational opportunities, and compromised surface water aesthetics.

Section 2.3.2 summarizes studies of aquatic organism and ecosystem impacts associated with excessive sediment levels in surface waters. These studies are not specific to sediment discharges from construction sites; however, given the wide range of characteristics associated with construction site sediments, they are likely to exhibit a range of behaviors and impacts similar to those of sediment from other pollutant sources.

2.3 Receiving Water Impacts of Sediment

2.3.1 Sediment and Turbidity Impaired Waters in the United States

When additional sediment enters a surface water, it can raise the TSS, TDS, and turbidity levels in the water column, increase deposition of sediment on the water body's bed, and impact the surface water's overall quality. Sediments, whether suspended, dissolved, or settled, play an important role in an ecosystem, providing habitat for benthic species, spawning grounds for fish, nutrient transport, and other roles. Appropriate levels of sediment are water body-specific, as sediment levels vary naturally among different types of water bodies and according to the geology, topography, stream gradient, water body morphology, vegetative land cover, climate, soil erodibility and other landscape characteristics of the contributing watershed (USEPA 2006b). Changing the quantity and behavior of sediments in an aquatic ecosystem, however, can have adverse consequences for its members (USEPA 2006b). Elevated sediment levels also negatively impact human use of aquatic resources for recreation, drinking water supply, irrigation, and navigation and can impair stormwater management systems and surface water aesthetics.

State Sediment Criteria

Many states have developed criteria to designate appropriate sediment levels for either all waters or for specific surface waters within the state. Many of these criteria are narrative statements describing the general nature of healthy sediment levels without attempting to provide numeric guidelines. Some states have also set numeric criteria for TSS and/or turbidity levels. In addition, specific Total Maximum Daily Loads (TMDLs) developed for water bodies within a state that are impaired for sediment are likely to contain numeric sediment standards for that water body.

Table 2-7 summarizes state sediment criteria for surface water quality. States using narrative criteria are noted with a "Yes" in the "Narrative TSS" or the "Narrative Turbidity" column.

Table 2-7: Sediment Criteria for Surface Water Quality, by State

State	Numeric TSS (mg/L)	Narrative TSS	Numeric Turbidity (NTU)	Narrative Turbidity
Alabama	-	-	50	-
Alaska	-	-	5-25[g]	Yes
Arizona	-	-	10-50[g]	Yes
Arkansas	-	-	10-75[g]	Yes
California	-	-	-	-
Colorado	-	Yes	-	-
Connecticut	-	-	-	-
Delaware	-	-	10	-
Florida	-	-	29	-
Georgia	-	Yes	-	Yes
Hawaii	10-55[a]	Yes	[a]	Yes
Idaho	-	Yes	-	Yes
Illinois	1000[b]	-	-	-
Indiana	-	Yes	-	Yes
Iowa	-	Yes	25	Yes
Kansas	-	Yes	-	Yes
Kentucky	-	Yes	-	Yes
Louisiana	-	Yes	25-50	Yes

Table 2-7: Sediment Criteria for Surface Water Quality, by State

State	Numeric TSS (mg/L)	Narrative TSS	Numeric Turbidity (NTU)	Narrative Turbidity
Maine	-	-	-	-
Maryland	-	-	150 at any time; 50 monthly average	Yes
Massachusetts	-	Yes	-	Yes
Michigan	-	-	-	-
Minnesota	-	-	-	-
Mississippi	-	Yes	50	Yes
Missouri	-	Yes	-	Yes
Montana	-	Yes	-	Yes
Nebraska	-	-	-	-
Nevada	25[c]	Yes	-	Yes
New Hampshire	-	Yes	10	Yes
New Jersey	25	-	30-50 at any time; 10-15 monthly average[g]	-
New Mexico	-	Yes	-	Yes
New York	-	Yes	-	-
North Carolina	500[d]	Yes	10-50[g]	-
North Dakota	30	-	-	-
Ohio	1-7.25[e,f]	-	-	-
Oklahoma	-	-	-	-
Oregon	-	Yes	10% increase above background	Yes
Pennsylvania	-	Yes	40-100 NTU[h]	Yes
Puerto Rico	-	-	-	-
Rhode Island	-	-	5-10 NTU[g]	-
South Carolina		Yes	10% above background[i]	Yes
South Dakota	53-263 (at any time); 30-150 (monthly average)[g]	Yes	-	Yes
Tennessee	-	Yes	-	Yes
Texas	-	Yes	-	Yes
Utah	35-90[g]	Yes	10-15 NTU[g]	Yes
Vermont	-	-	10-25 NTU[g]	-
Virginia	-	Yes	-	Yes
Washington	-	-	5-10 NTU or 10-20% increase[g]	-
West Virginia	-	-	10 NTU or 10% increase	-
Wisconsin	-	Yes	-	Yes
Wyoming	-	Yes	10-15 NTU[g]	Yes

Table 2-7: Sediment Criteria for Surface Water Quality, by State

State	Numeric TSS (mg/L)	Narrative TSS	Numeric Turbidity (NTU)	Narrative Turbidity

a. Varies based on several parameters
b. Individual water body criteria from Illinois EPA (2006)
c. Individual water body criteria from Nevada DEP (2003)
d. Standard is for total dissolved solids (TDS)
e. Individual water body criteria from Ohio EPA (2000)
f. Individual water body criteria from Ohio EPA (2005)
g. Varies based on water body classification
h. Specific to the Neshaminy Creek Basin (USEPA 2006b)
i. Specific to trout stock waters

Sources: USEPA (2006b); Illinois EPA (2006); Nevada DEP (2006); Ohio EPA (2000); Ohio EPA (2005)

Sediment and Clean Water Act (CWA) Sections 303(d) and 305(b) Impaired Waters

Under Section 305(b), states are required to submit to EPA every two years a report describing the water quality of all waters in the state. States have determined the appropriate uses of water bodies within their jurisdiction. Typical uses include recreation, drinking water supply, navigation, and wildlife habitat, among others. If a water body fails to meet any one of its designated uses, Section 305(b) of the CWA requires states to list that water body as "impaired." If the water body currently meets its designated uses but may not in the future, the state may list that water body as "threatened." Along with this list, states are required to submit under Section 303(d) of the CWA a list of all impaired and threatened water bodies still requiring Total Maximum Daily Load (TMDL) criteria (USEPA 2005a).

EPA's 2002 *National Water Quality Inventory: Report to Congress* (USEPA 2007d), which summarizes the states' 305(b) reports, lists "sediment/siltation" as the most prevalent cause of pollution in rivers and streams in terms of number of miles affected and the fourth most prevalent for lakes, ponds, and reservoirs. *Table 2-8* and *Table 2-9* present impairment of water bodies from the 2002 305(b) reporting cycle (the most recent year for which data are publicly available). It is important to note that the water bodies assessed do not represent all water bodies in the United States. The aforementioned report indicates that only 19 percent of rivers and streams; 37 percent of lakes, ponds, and reservoirs; and 35 percent of estuaries were evaluated for the 2002 reporting cycle (USEPA 2007d).

Sediment-related impairment in the 305(b) data falls into several categories of pollutant causes, including "sediment/siltation," "turbidity," and "suspended solids." According to documentation for WATERS, "sediment/siltation" impairment is related to settled sediments, while "turbidity" impairment is caused by suspended particles, and this parent cause includes "suspended solids."

Of the assessed rivers and streams, more than 103,000 miles were reported as impaired due to "sediment/siltation," along with nearly 1,264,000 acres of lakes, ponds, and reservoirs. "Turbidity" and "suspended solids" impair close to 29,500 miles of rivers and streams, and nearly 839,000 acres of lakes, ponds, and reservoirs. Nearly 119 square miles of bays and estuaries are reported as impaired by "sediment/siltation" and 1,689 square miles by "turbidity" or "suspended solids."

Table 2-8: "Sediment/Siltation" Impairment in 305(b)-Assessed Waters [1]

EPA Region	River, Stream (miles)		Lake, [2] Pond, Reservoir (acres)		Bay, Estuary (sq. miles)	
	Assessed	Impaired	Assessed	Impaired	Assessed	Impaired
1	21,699.5	5,276.8	1,491,595.5	175,527.5	1,018.7	0.1
2	12,814.8	4,874.3	409,319.9	50,960.0	1,675.9	0.5
3	127,468.7	7,541.4	271,042.5	44,378.5	4,996.1	35.2
4	66,507.3	6,766.5	1,173,482.6	19,554.0	329.8	0.0
5	128,144.0	22,942.4	3,699,833.3	155,925.9	86.0	0.0
6	49,969.2	3,514.5	2,726,913.2	179,666.7	7,802.7	6.0
7	56,467.7	9,512.9	693,980.7	9,958.0	0.0	0.0
8	110,169.4	6,892.0	1,912,016.9	252,899.0	0.0	0.0
9	41,477.6	19,362.1	1,272,016.2	279,590.1	958.8	73.0
10	80,414.5	17,271.3	1,181,681.6	95,725.8	13,577.7	3.8
Total	**695,132.8**	**103,954.1**	**14,831,882.3**	**1,264,185.4**	**30,445.7**	**118.7**

1 Values are preliminary pending correction of EPA's 305(b) dataset, publicly available through the WATERS database, which links 305(b)-assessed waters to their causes of impairment. The 2002 305(b) report (USEPA 2007d) lists 100,446 miles of rivers and streams and 1,317,938 acres of lakes and reservoirs impaired by "sediment/siltation."

2 Does not include Great Lakes

Source: WATERS (USEPA 2008g)

Table 2-9: "Turbidity" and "Suspended Solids" Impairment in 305(b)-assessed Waters [1]

EPA Region	River, Stream (miles)		Lake [2], Pond, Reservoir (acres)		Bay, Estuary (sq. miles)	
	Assessed	Impaired	Assessed	Impaired	Assessed	Impaired
1	21,699.5	1,719.0	1,491,595.5	21,275.1	1,018.7	9.3
2	12,814.8	23.7	409,319.9	0.0	1,675.9	0.0
3	127,468.7	525.9	271,042.5	0.0	4,996.1	1,528.1
4	66,507.3	458.6	1,173,482.6	1,810.0	329.8	7.1
5	128,144.0	6,292.7	3,699,833.3	136,122.5	86.0	0.0
6	49,969.2	9,936.3	2,726,913.2	469,765.5	7,802.7	52.0
7	56,467.7	135.0	693,980.7	95,423.0	0.0	0.0
8	110,169.4	3,340.6	1,912,016.9	23,483.5	0.0	0.0
9	41,477.6	5,913.1	1,272,016.2	86,962.7	958.8	37.9
10	80,414.5	1,111.1	1,181,681.6	3,803.5	13,577.7	54.9
Total	**695,132.8**	**29,455.9**	**14,831,882.3**	**838,645.8**	**30,445.7**	**1,689.3**

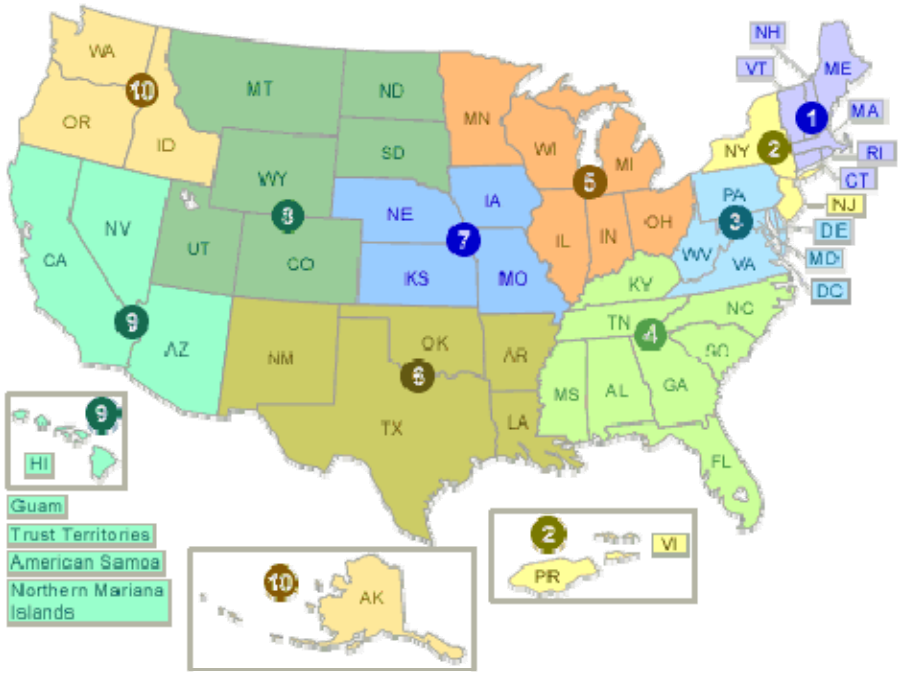
1 Values are preliminary pending correction of EPA's 305(b) dataset publicly available through the WATERS database, which links 305(b) assessed waters to their causes of impairment. Turbidity was not among the pollutants listed in the 2002 305(b) report (USEPA 2007d).

2 Does not include Great Lakes

Source: WATERS (USEPA 2008g)

Figure 2-2 presents the United States divided into EPA regions, which are referred to in the tables above and throughout the remainder of this document.

Figure 2-2: EPA Regions



Sediment is one of the most common pollutants affecting water bodies listed on state 303(d) lists. According to data from EPA's WATERS database (USEPA 2008g), 16 percent of all waters on 303(d) lists are impaired by sediment and 5.5 percent by turbidity. This amounts to 51,000 miles of rivers and streams and 562,000 acres of lakes, ponds, and reservoirs impaired by sediment. Turbidity impairs 16,000 miles of rivers and streams, and 165,000 acres of lakes, ponds, and reservoirs. It should be noted that the 303(d) lists only include waters that are in need of TMDL development, so the number of waters impaired by sediment pollution is more accurately represented by waters listed as impaired in the 305(b) assessment. However, the 303(d) data identify sources of pollutants, and construction is listed as the impairment source for 17,000 miles of rivers and streams and for more than 486,000 acres of lakes, ponds, and reservoirs, though only 25 states have a category for construction in their list of impairment sources. Given the general difficulties with ascribing a surface water's impairment to a specific source of pollution, particularly a transient pollutant source such as construction, it is possible that some states have not been able to verify construction activity as a source of surface water impairment. As with the 305(b) data, 303(d) lists are limited to assessed waters and thus exclude about 80 percent of rivers and streams, and more than 60 percent of lakes, ponds, and reservoirs. Thus, it is likely that the actual magnitude of sediment and turbidity impairment on both the 305(b) and 303(d) lists is more extensive than reflected by currently reported numbers.

Current Total Suspended Solid Concentrations in U.S. Rivers and Streams

TSS concentrations vary from water body to water body. EPA used the SPARROW model (see *Chapter 6, Section 6.1,* for more details) to model current TSS concentrations in the United States' river and stream network. TSS concentrations are monitored at USGS Stream Gauging stations, though data for these concentrations are not available for the entire RF1 network. SPARROW incorporates the available data on TSS concentrations into its modeling process and predicts TSS concentrations for all reaches in the RF1 network. Due to data limitations, EPA did not model current turbidity levels. *Table 2-10*

summarizes the distribution of current TSS concentrations in the Reach File Version 1.0 (RF1)[1] network as estimated by EPA's analysis. The RF1 network consists of approximately 650,000 miles of rivers and streams and associated lakes and reservoirs in the conterminous 48 states. The median TSS concentration as estimated by this analysis is 261.91 mg/L when weighted by reach length. The range of TSS concentrations falls between 24.79 mg/L and 3,203.47 mg/L at the 5th and 95th percentiles respectively. The average TSS concentration is 704.88 mg/L, indicating that concentrations in the 90th to 95th percentile range are high enough to shift the average concentration significantly higher than the median concentration.

Table 2-10: SPARROW Distribution of TSS Concentrations[1, 2]

RF1 Reach Count	RF1 River 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	704.88	24.79	92.71	261.91	768.51	3,203.47

1 Concentrations weighted by reach length. Statistics exclude potential outliers (defined as values above the 95th percentile (TSS = 6,049.7 mg/L)).

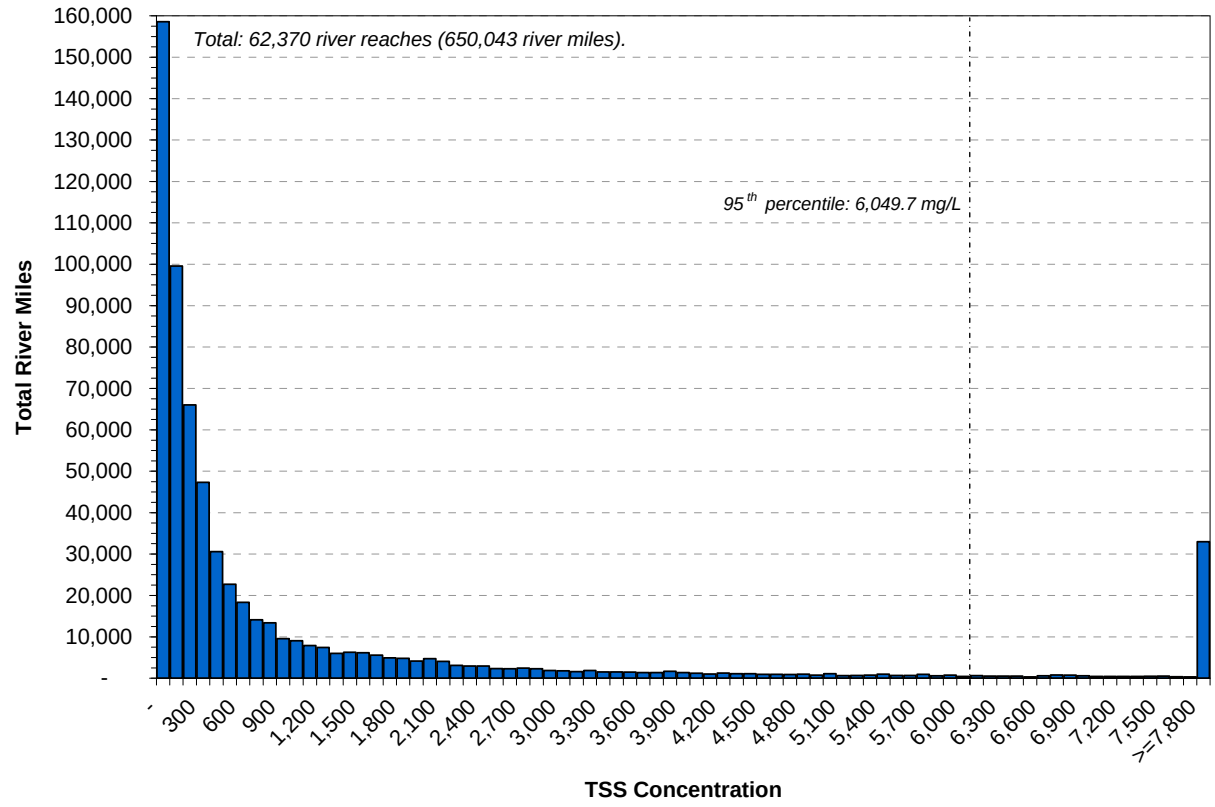
2 Results are reflective of both polluted and unpolluted waters.

Source: USEPA analysis

Figure 2-3 shows total RF1 river 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 concentrations up to and exceeding 7,800 mg/L can be found in some U.S. waters.

[1] The Reach File Version 1.0 (RF1) database contains location, length, and other information for 62,370 reaches in the conterminous United States (USEPA 2007d). Each individual reach is given a unique identifier known as an RF1 ID, which is used throughout this Environmental Assessment to identify individual waters benefitting from water quality improvements expected form this regulation. See the RF1 documentation for more information at http://www.epa.gov/waters/doc/rf1_meta.html.

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



Source: USEPA analysis

A certain level of sediment is present in surface waters under natural, undisturbed conditions, and some level of sediment is necessary for the natural biological function of most surface waters. Current levels of sediment discharge to the United States surface water network are much higher, however, than would be observed under natural, undisturbed conditions due to sediment discharges from construction, agriculture, eroding streambeds due to land use change, and other human activities.

Sediment from construction sites (and other sources) can have a wide range of effects on aquatic ecosystems located downstream of those construction sites. *Section 2.2.2* discusses the types of impacts sediments can have on aquatic life. *Chapter 4* and *Chapters 7-10* describe the impacts sediments can have on human use of aquatic resources, including aquatic life.

2.3.2 Aquatic Life Impacts of Sediment

Increased levels of sediment in surface waters have been shown to have negative impacts on aquatic life. These effects are documented for primary producers, such as aquatic plants, as well as for mollusks, benthic macroinvertebrates, fish, and other aquatic organisms. The various manifestations of elevated sediment levels (elevated TSS, TDS, turbidity, and sedimentation levels) affect different aquatic organisms in different ways. Some organisms are directly impacted. Other organisms are harmed indirectly as direct impacts on some organisms cascade upward or downward through the ecosystem. For example, aquatic vegetation reduces photosynthesis and growth under high turbidity levels and resulting

low light conditions. Reduced vegetation growth can reduce food availability for a variety of aquatic organisms, depressing their survival rates (Henley et al. 2000).

Sediment impacts have been researched since at least the 1960s through both laboratory dose-response and in-stream field studies, though many aspects remain to be explored. Much research has been done on the direct effects of sediment on fish, particularly salmon because of their economic importance. Many more recent studies address the interconnected nature of aquatic ecosystems and examine a broader range of organism types. Documented effects and relevant studies are organized in the discussion below by the type of aquatic organism studied (primary producer, invertebrate, fish) and by the type of organism reaction. A section specific to threatened and endangered species follows this discussion.

Primary Producers

Primary producers such as algae, phytoplankton, and submerged aquatic vegetation play a critical role in the health of aquatic ecosystems by transforming sunlight into energy and providing food for other aquatic organisms. Their existence and productivity is therefore crucial to the survival of the entire ecosystem. A compounding factor of reduced quantities of aquatic macrophytes and algae due to sediment impacts in an aquatic ecosystem is that there are fewer stems, leaves, and other materials to slow water flow and dislodge suspended sediment from the water column. This effect makes recovery more difficult and allows elevated TSS concentrations to persist further downstream (Wood and Armitage 1997).

A 1997 study by Wood and Armitage identifies four primary ways in which elevated sediment levels can affect primary producers in aquatic ecosystems:

- Reduce light penetration,, thus reducing photosynthesis levels
- Damage aquatic vegetation leaves and stems through abrasion
- Prevent algal cells from attaching to water body substrate
- Reduce organic content of periphytic cells.

Light reduction: A reduction in primary production as a result of increased turbidity due to elevated TSS concentrations is the most common impact that has been documented by LaPerriere et al. (1983), Lloyd (1987), and Rivier and Seguier (1985) (all as cited in Waters 1995), Edwards (1969) and Meyer and Heritage (1941) (both as cited in Kerr 1995). LaPerriere et al. (1983) find that a moderately turbid stream with turbidity measurements of approximately 170 NTU had oxygen production reduced by 0.08–0.64 g/m^2 per day. In highly turbid streams exceeding 1,000 NTU, primary production was undetectable. Lloyd (1987) develops a model relating turbidity and primary production. The model indicates that in a clear, shallow stream, a turbidity increase of 5 NTU decreases primary production between 3 percent and 13 percent, and an increase of 25 NTU reduces primary production by up to 50 percent (Waters 1995). The turbidity levels recorded by studies of construction sites often exceed 100 NTU and are high enough to cause a reduction in primary production. A study by Guenther and Bozelli (2004) attempts to determine the ultimate cause of reduced photosynthetic activity in turbid waters, which they postulate could be a result of either lower light levels or of increased sinking of phytoplankton due to their bonding to sediment particles. The study's results suggest that reduced light penetration is the primary cause of lower primary production.

Physical damage: Algae and aquatic macrophytes can also be physically damaged by elevated TSS concentrations. This effect is noted by Lewis (1973), cited in Kerr (1995), from an experiment in which concentrations in excess of 100 milligrams per liter of suspended coal particles are shown to cause 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.

Reduced organic periphyton: Periphyton consist of the collection of aquatic organisms, bacteria, and dead organic matter that are attached to submerged surfaces in aquatic ecosystems. A reduction or complete elimination of periphytic cells 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). This impact was manifested by a general reduction in the number and diversity of aquatic primary producers. 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. Herbivores consuming the periphyton also consume the elevated quantities of sediment, reducing the nutritional benefit from their feeding (Ryan 1991).

Invertebrates

Aquatic invertebrates, including zooplankton, aquatic insects, and bivalves (mussels and clams), are an important link in the aquatic food chain between primary producers and larger animals such as fish (Waters 1995). The effects of elevated sediment levels on invertebrates, particularly on benthic macroinvertebrates, have been the focus of a large number of studies over the past several decades. The impacts of sediment on invertebrates can be classified into the following categories identified by Kerr (1995) and Wood and Armitage (1997):

- Loss of habitat
- Altered movement and increased drifting
- Altered species composition
- Physiological changes
- Reduced feeding activity
- Reduced growth rates
- Toxicity and direct mortality

Loss of habitat: Increased sediment levels in water bodies can degrade benthic macroinvertebrate habitat. Sediment settles from the water column onto the substrate to which sedentary organisms attach and also fills small nooks and crevices in water body beds, known as interstitial spaces, where a variety of species live, reproduce, and seek protection from predators. Siltation of previously coarse substrates is implicated in the reduction of mussel populations by Ellis (1936) and Marking and Bills (1980), both as cited in Waters (1995). Waters (1995) also cites a severe sedimentation case documented by Cordone and Pennoyer (1960) in which "bottom fauna densities and biomass…were reduced to less than 10% of former levels" after a "gravel-washing operation covered stream bottoms up to one foot in depth or more and reduced the substrate to a 'hard, bedrock-like' condition." Ryan (1991), as cited in Henley et al. (2000), concludes that as little as a 12 percent to 17 percent increase in interstitial fine sediment could result in a 16 percent to 40 percent decrease in macroinvertebrate abundance.

Altered movement and increased drifting: Movement downstream (drifting) of benthic macroinvertebrates has been found to increase with elevated sediment concentrations (Kerr 1995). Downstream drift reduces organism abundance in the impacted surface water. Increases in invertebrate drift have been shown to be specifically related to the turbidity caused by suspended sediments (Waters

1995). White and Gammon (1977), as cited in Waters (1995), postulate that prolonged high concentrations of suspended sediment could reduce benthic macroinvertebrate population levels.

Altered species composition: Loss of habitat and increased drifting can culminate in a change in the species composition of an aquatic community as sediment-sensitive species are replaced by sediment-tolerant species. Gammon (1970), as cited in Kerr (1995), finds that TSS concentrations between 40 mg/L and 120 mg/L reduced macroinvertebrate density by 25 percent, and greater TSS concentrations caused more substantial reductions of 60 percent. Organisms intolerant of turbid waters include mayflies, clams, and bryozoans (Cooper (1987), as cited in Kerr (1995)). Sediment discharges from construction activity are implicated in reduced invertebrate abundance and modified community composition in several studies, including Chisholm and Downs (1978), Cline et al. (1982), Ohio EPA (2003a), and Levine et al. (2003).

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 or digestive system (Kerr 1995), resulting in injury to or death of the organism.

Reduced feeding activity: Another effect of elevated TSS and turbidity levels on aquatic invertebrates is a reduction in their feeding activity. Kerr (1995) notes that this impact is predominant in filter feeding species such as mussels and clams and cites Ellis (1936), which finds that under turbid conditions, mussels and clams reject silt-laden food and, under highly turbid conditions, close their shells completely. Waters (1995) also discusses sediment's impact on feeding, citing Ellis (1936) and Aldridge (1987), who find that sediment additions impaired feeding in three species of bivalves. Reduced feeding activity can result in impaired functioning or death of organisms.

Reduced growth rates: Kerr (1995) cites three studies (Appleby and Scarratt (1989), Kirk (1992), and Paffenhofer (1972)) that find reduced growth rates in invertebrates subjected to elevated sediment levels due, potentially, to reduced feeding activity or reduced food value.

Toxicity and direct mortality: Sediment can directly or indirectly increase invertebrate mortality levels. Forbes et al. (1981), as cited in Kerr (1995), finds that suspended sediment are acutely toxic to amphipods (shrimp-like crustaceans). Henley et al. (2000) concludes 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. The review also suggests that many of the more than 30 extinct species of North American mussels died out due to a loss of habitat, most likely from sedimentation of the substrate (Henley et al. 2000).

The overall effect of the various impacts from elevated sediment levels on aquatic invertebrates is a general reduction in their abundance, as documented in the literature reviews conducted by Kerr (1995), Waters (1995), and Henley et al. (2000). Studies of specific construction sites also find that macroinvertebrate communities were impaired downstream of construction sites, citing elevated levels of TSS and turbidity as the causes (Chisholm and Downs 1978; Cline et al. 1982; Fossati et al. 2001; Levine et al. 2003, 2005; Selbig et al. 2004; Ohio EPA 1996a, 1997f, 1998d, 1999d, 2006b).

Fish

A substantial amount of research has been done to identify and quantify the effects of elevated sediment levels on a variety of fish species. The observed impacts generally fall into the following categories:

- Avoidance
- Feeding and hunting

- Breeding and egg survival
- Juvenile survival
- Physical damage
- Mortality and population level changes

Avoidance: Because fish are much more mobile than aquatic plants, plankton, or invertebrates, they are capable of actively avoiding waters with high levels of suspended sediment and/or turbidity if migration routes to alternative habitats are available. Barton (1977) notes a relocation of fish populations away from areas with elevated TSS levels in a small stream in southern Canada. Servizi and Martens (1987, as cited in Reid et al. 2003) determine that turbidity levels in excess of 37 NTU elicit avoidance behavior among fish populations. These types of avoidance behaviors may reduce or eliminate fish populations in stream reaches with sustained elevated sediment levels.

Feeding and hunting: Elevated TSS and turbidity levels have been shown to reduce feeding rates in fish through studies on bluegills, arctic grayling, Coho salmon, and steelhead (Kerr 1995). This effect is generally believed to be a result of decreased visibility caused by turbidity, as many fish rely on their vision to locate food or prey. De Robertis et al. (2003) find support for this theory in a study where piscivorous fish, which visually locate their prey at longer distances, were more severely affected by increases in turbidity levels than were planktivorous fish, which locate their prey at much shorter distances. Research into the impacts of elevated sediment levels by USEPA (2004a) concludes that turbidities greater than 25 NTU or TSS levels of 2,000–3,000 mg/L or greater are sufficient to reduce the predation abilities of fish. This reduction in predation can benefit prey species that use turbid water to hide from predators (Rowe et al. 2003). However, a paper by the Canadian Department of Fisheries and Oceans (2000) suggests that as turbidity levels continue to increase, the advantage of reduced predation is outweighed by the prey fishes' own increased difficulty in locating food. A compounding effect of elevated sediment levels is decreased food availability due to invertebrate population decline and decreased primary productivity, as described in the sections above.

Breeding and egg survival: In addition to reducing habitat for invertebrates, elevated sediment levels also reduce spawning habitat for multiple fish species. Saunders and Smith (1965, as cited in Kerr 1995) find that increased TSS levels reduced fish spawning activity. Other studies document lower survival rates for eggs spawned in surface waters impacted by elevated TSS and turbidity levels (Chapman 1988 as cited in Canadian Department of Fisheries and Oceans 2000). This effect is most likely due to the fact that sediments, particularly the finer fractions, can reduce the level of oxygen available to eggs deposited in the substrate (Kerr 1995). Bjornn et al. (1977), as cited in Kerr (1995), find that successful emergence of fish fry from eggs begins to decline when fine sediment levels in the substrate are in excess of 20–30 percent. Reynolds (1989, as cited in Henley et al. 2000) finds that increases in turbidity levels are associated with increases in sac fry mortality in arctic grayling. These reductions in spawning and survival of egg and fry have the potential to reduce fish population levels in surface waters impacted by sediments.

Juvenile survival: Research into the effects of sediment on fish populations has noted that high sediment concentrations can impact juvenile fish more severely than adults. Juvenile Coho and Chinook salmon exhibited abnormal surfacing behavior under higher sediment concentrations, rendering them more vulnerable to avian predators (Canadian Department of Fisheries and Oceans 2000). The Canadian Department of Fisheries and Oceans (2000) study also notes that excessive sediment levels can act as a stressor to juvenile fish and consequently increase their predation risk. As with a reduction in fish egg and

fry survival, reduced survival of juvenile fish to adulthood has the potential to reduce fish population levels.

Physical damage: Sediment particles, particularly more angular particles, are capable of causing physical damage to fish by clogging gills and impairing breathing or through 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 their skin and gills, but this reaction stresses fish. Reid et al. (2003) finds that elevated sediment levels reduces the ability of rainbow trout to withstand hypoxic conditions and thereby reduced their average life span.

Mortality and population level changes: The potential cumulative effect of these different impacts from elevated sediment levels is a decrease in fish population levels in impacted areas. Reductions can take place through direct mortality in the short term and through reduced reproductive success in the long term. Newcombe (1994) and Newcombe and Jensen (1996), (both cited in Henley et al. 2000) find that elevated sediment concentrations are associated with increased mortality in at least 14 species of fish. 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 northeast Missouri. Barton (1977), Graves and Burns (1970), and Ohio EPA (1999c) find that standing fish crops were reduced immediately downstream from construction activity.

The severity of impacts on fish populations is related to the intensity and duration of elevated sediment levels (Anderson et al. 1996, as cited in USEPA 2004a). Short-term increases in TSS and turbidity levels can occur during spring thaws, storms, and other natural events. However, a construction project may produce sediment levels significantly higher those associated with natural events, for longer periods of time, and/or at times when an aquatic ecosystem is unaccustomed to receiving such sediment inputs, exposing fish populations to the impacts described above.

Threatened and Endangered Species

Threatened and endangered (T&E) and other special status species can be adversely affected in several ways by construction activities. This section discusses the potential for impacts from construction site sediment discharges on aquatic T&E species. Construction activity can jeopardize aquatic T&E species through increased sediment levels, a significant contributor to aquatic plant, invertebrates, and fish injury and mortality as discussed in the section above. For example, elevated sediment concentrations adversely affect aquatic plants and microscopic primary producers through reduced light penetration, which impedes photosynthesis (Kerr 1995). Elevated sediment levels impact aquatic invertebrates and fish through habitat modification and destruction as well as through direct adverse physical impacts on species. Some fish species cannot survive in water with high levels of suspended sediments, and sediments that settle on river or stream bottoms can smother fish egg deposits (NOAA 2004). 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). The departments are also referred to collectively herein as the Services. The USFWS is responsible for terrestrial and freshwater species (including plants) and migratory birds, whereas the NMFS deals with marine species and anadromous fish (USFWS 2008a). Various state agencies and departments have state-level jurisdiction over T&E species.

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. Species are selected for listing based on petitions, surveys by the Services or other agencies, and other substantiated reports or field studies. 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.

This discussion focuses on federally listed T&E species based on information in the USFWS Threatened and Endangered Species System (TESS) database (USFWS 2008b), 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 2008) at http://www.natureserve.org/explorer/. Additional information on state-listed species is available from state T&E coordinators.

Numerous physical and biological stressors have resulted in the listing of multiple aquatic species as threatened or endangered. The major factors include habitat destruction or modification, displacement of populations by exotic species, dam building and impoundments, increased sediment and turbidity levels in the water column, sedimentation, various point and nonpoint sources of pollution, poaching, and accidental catching.

As discussed in the sections above and in *Chapter* XX*3*XX, aquatic organisms are susceptible to adverse effects from pollutant discharges from construction activity. Increased sedimentation, TSS, and turbidity levels in water bodies downstream of construction activity can depress organism growth, survival, and reproduction, thereby leading to further decline in an impacted T&E species population.

Because construction activities subject to NPDES permitting occur in every state, 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, clams, crustaceans, and corals (USFWS 2008b). For a complete list of threatened and endangered aquatic species, see *Appendix A.* 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 sedimentation may have adverse impacts on species that are already in peril either directly or indirectly through impacts on supporting food chains (Cloud 2004; NatureServe 2008; USFWS 2008b).

Several T&E species are thought to be particularly susceptible to excessive sedimentation. 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 2008). Other similarly sediment-susceptible species of mollusks include the oyster mussel, purple bankclimber, Louisiana Pearlshell, Alabama Heelsplitter, and Ouachita rock pocketbook (USFWS 2008b).

Many fish species are also vulnerable to sedimentation. Sedimentation is a risk factor for a number of shiner species, including the Arkansas river shiner, Pecos bluntnose shiner, and blue shiner (NatureServe 2008). 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 2008). Sedimentation is also listed as a threat factor to a number of minnow and darter species (e.g., loach minnow, spikedace, and leopard darter) (Cloud 2004; NatureServe 2008; USFWS 2008b). Sedimentation has also contributed to the threatened status of some populations of rainbow trout and several salmon species in the Pacific Northwest (NatureServe 2008). California coho salmon and steelhead trout are listed and 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).

3 Other Pollutants and Impacts from Construction Discharges

In addition to mobilizing and discharging sediment, settleable solids, TDS, TSS, and turbidity to surface waters, construction activity has the potential to contribute many other kinds of pollutants to adjacent surface waters. These pollutants, though not as well studied as sediment in the construction context, have the potential to impact water quality and therefore aquatic life and human use of water resources.

Construction activity involves the use of a wide variety of materials with the potential to produce metal, wood, plastic, and other debris that can be washed into nearby waters if not properly managed. In addition, chemicals associated with construction materials and equipment such as wood treated with creosote and chromated copper arsenate, paint, adhesives, solvents and vehicle oils, fuel, and grease can leach or wash from equipment and materials during rain events and subsequently discharge to surface waters. Asphalt and concrete pavement and building materials contain chemicals that can alter water pH or have toxic effects in aquatic environments. Concrete wastes have an elevated pH due to the lime they contain. In addition, during the manufacturing process, concrete is often mixed with industrial wastes that can elevate metal levels in concrete waste. Asphalt materials contain a variety of organic compounds, including polycyclic aromatic hydrocarbons (PAHs) and additives whose environmental effects have not been thoroughly characterized. Coal tar-based sealcoat for driveway and parking lot surfaces, in particular, contains very high levels of PAHs (Van Metre et al. 2006) and has been documented as a cause of a fish kill (Montgomery County 2000). Commonly used landscaping materials such as fertilizer, mulch, lime, and pesticides can also wash from construction sites into surface waters, contributing nutrients, oxygen-demanding material, and toxic pollutants, including metals present in many fertilizers and pesticide residues.

In addition to pollutant discharges from construction equipment and materials, land disturbance associated with construction can mobilize pollutants already present on a site when construction begins, These pollutants can be natural in origin (e.g., nutrients, oxygen-demanding materials, or metals naturally present in a site's soil), or they can derive from earlier contamination of a site by human activity. Former industrial activities can deposit a wide range of pollutants in a site's soil, including toxic metals and organic compounds. Land previously used for agriculture can contain high levels of pesticide residues, nutrients from fertilizers, or salts from irrigation.

Pollutants other than sediment have not been as widely researched or documented in the construction environment. This is despite the fact that many of the pollutants described above adsorb to or otherwise travel with sediment discharges. EPA was able to locate 11 studies and reports that describe pollutants other than sediment found in construction site discharges or at elevated levels immediately downstream of construction sites. The more frequently described pollutants are:

- Nutrients (nitrogen and phosphorus compounds)
- Polycyclic aromatic hydrocarbons (PAHs)
- Metals

These pollutants are discussed in additional detail in the sections below. Other pollutants such as ammonia (Ohio EPA 1997f), fecal coliform (Ohio EPA 1997f, 1998c, 1998d), and chemicals from spills were also described in some studies. In general, however, they do not seem to be well characterized.

An additional impact from construction activity is that vegetation removal, soil compaction, and modified surface gradients can increase the volume and intensity of water runoff discharging from a construction site. 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 the periodic elevated flow volumes (Wheeler et al. 2003). The changes in streambed morphology can increase sediment erosion, destabilize stream banks, elevate sediment levels in downstream waters, and cause long-term changes in aquatic and riparian habitat. These changes can result in the destruction of previously existing aquatic communities.

3.1 Documented Pollutants Other than Sediment from Construction Sites

3.1.1 Nutrients

Nutrients, primarily phosphorus and nitrogen compounds, have the potential to discharge from construction sites in stormwater discharges, elevating nutrient levels in receiving waters. Nitrogen is typically found in soluble forms. Phosphorus may be released in either soluble form or attached to sediment (Daniel et al. 1979). Three of the studies discussed in *Chapter 2* measured nutrient as well as sediment levels in impacted surface waters. Two additional papers discuss nutrients in construction site discharges qualitatively. *Table 3-1* lists these studies and the specific nutrients they describe.

Table 3-1: Nutrients in Studies of Construction Site Discharges

Author(s)	Year	Study Name	Specific Nutrients
Daniel et al.	1979	Sediment and Nutrient Yield from Residential Construction Sites	Phosphorus, Nitrogen
Novotny and Chesters	1989	Delivery of Sediment and Pollutants from Nonpoint Sources: A Water Quality Perspective	Phosphorus
Harbor et al.	1995	Reducing Nonpoint Source Pollution from Construction Sites Using Rapid Seeding and Mulching	Phosphorus
Ohio EPA	1997	Biological and Water Quality Study of Twin Creek and Selected Tributaries 1995	Nitrate, Nitrite, Ammonia
Fossati et al.	2001	Impact of Sediment Releases on Water Chemistry and Macro-invertebrate Communities in Clear Water Andean Streams (Bolivia)	Phosphates

Daniel et al. (1979) reported that elevated nitrogen and phosphorus loadings in waters receiving construction site discharges were 90 and 99 percent (respectively) explained by elevation in sediment loadings from residential construction sites sampled in Wisconsin. Novotny and Chesters (1989) also studied construction sites in Wisconsin and concluded that nutrient discharges are closely related to sediment discharge levels because water discharging from construction sites carries both soil and the nutrients it contains. Harbor et al. (1995) corroborated this conclusion, and also noted that certain erosion control measures reduced sediment discharge by up to 86 percent and phosphorus discharge by up to 80 percent.

Though Ohio EPA (1997f) and Fossati et al. (2001) do not include detailed discussions of nutrient levels in sediment at construction sites, they do note that increased levels of nutrients have accompanied increases in sediment levels immediately downstream of construction activity. Ohio EPA (1997f) directly attributes these increases to the lack of proper construction site discharge management at a golf course. Elevated levels of TSS, nitrogen compounds (including ammonia, which has both nutrient and toxic

properties), and fecal coliform were recorded at a chemical sampling site about half of a mile downstream of the construction discharge site.

3.1.2 Polycyclic Aromatic Hydrocarbons

Polycyclic aromatic hydrocarbons (PAHs) are not frequently documented as construction site discharge pollutants. However, Ohio EPA water quality reports recognize construction sites, specifically road construction sites, as possible contributors to elevated 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 the asphalt and tar used in road construction and repair, and it is possible for PAHs to leach into sediment and construction site discharges during rain events (Ohio EPA 1998d).

3.1.3 Metals

Like PAHs, metals have not been frequently documented as pollutants in construction site discharges. However, an Ohio EPA report on Raccoon Creek attributes increased metal concentrations to pressure-treated wood products used at a nearby residential construction site (Ohio EPA 1997d). Barrett et al. (1995) recorded concentrations of iron and zinc increasing much more than those of other metals as TSS concentrations increased in surface waters due to construction site discharges. The study found that TSS concentrations explained 40 to 94 percent of the variance in iron concentration (depending on the measurement site), but identified no direct correlation to other parameters and did not hypothesize as to the ultimate source of the iron or zinc increases.

3.2 Receiving Water and Aquatic Life Impacts of Pollutants Other than Sediment

3.2.1 Nutrients

Nutrients are an impairment source for a large number of surface waters in the United States, ranking just behind sediment in total number of listings. Nutrient impairment (including associated algal growth and algal blooms) impairs the second-largest number of stream and river miles and the fifth-largest number of lake, pond, and reservoir acres. *Table 3-2* summarizes the total number of stream and river miles and lake, pond, and reservoir acres reporting nutrient or algal-related impairment in the 2002 305(b) assessment data. This information was obtained from the EPA WATERS database (USEPA 2008g). As with sediment listings, it should be noted that the 305(b) data only assess 19 percent of rivers and streams; 37 percent of lakes, ponds, and reservoirs; and 35 percent of bays and estuaries, so the actual number of waters impaired by nutrient pollution is likely to be much higher than the 2002 305(b) data indicate.

As shown in *Table 3-2*, nutrients impair more than 21,000 miles of rivers and streams; more than 1.7 million acres of lakes, ponds, and reservoirs; and more than 1,800 square miles of bays and estuaries. Region 5 shows the greatest length of assessed rivers and streams impaired, as well as the largest number of impaired lake, pond, and reservoir acres. Region 3, which includes the Chesapeake Bay, accounts for almost all nutrient-impaired bay and estuary area in the United States, with more than 1,800 square miles. According to WATERS, nutrients are also the cause of impairment for 13 percent of 303(d)-listed surface waters (USEPA 2008g), the more severely impaired waters requiring TMDL development.

Table 3-2: Nutrient Impairment in 305(b)-Assessed Waters[1]

EPA Region	River, Stream (miles)		Lake,[2] Pond, Reservoir (acres)		Bay, Estuary (sq. miles)	
	Assessed	Impaired	Assessed	Impaired	Assessed	Impaired
1	21,699.5	3,080.1	1,491,595.5	210,600.4	1,018.7	0.0
2	12,814.8	0.0	409,319.9	140.8	1,675.9	0.0
3	127,468.7	3,447.2	271,042.5	41,053.4	4,996.1	1,831.0
4	66,507.3	1,525.6	1,173,482.6	23,414.0	329.8	0.0
5	128,144.0	9,315.6	3,699,833.3	964,737.0	86.0	0.0
6	49,969.2	873.8	2,726,913.2	90,150.4	7,802.7	6.0
7	56,467.7	784.6	693,980.7	89,630.0	0.0	0.0
8	110,169.4	858.7	1,912,016.9	265,414.6	0.0	0.0
9	41,477.6	269.7	1,272,016.2	35,778.6	958.8	7.8
10	80,414.5	1,525.0	1,181,681.6	71.7	13,577.7	0.0
Total	**695,132.8**	**21,680.1**	**14,831,882.3**	**1,720,990.9**	**30,445.7**	**1,844.7**

1 Values are preliminary pending correction of EPA's publically available 305(b) dataset
2 Does not include Great Lakes

Source: WATERS (USEPA 2008g)

While nutrients are necessary for the primary production (photosynthetic activity) that forms a fundamental part of the aquatic food chain, nutrients are also of concern as a pollutant in surface waters because they can cause eutrophication. Eutrophication is a process by which excessive nutrient levels increase the growth rates of primary producers, particularly algae, which can drastically alter a water body's ecological balance and also impair human use of the water body.

Excessive algal growth rates can lead to algal blooms, certain types of which can be both directly and indirectly harmful to both aquatic organisms and human beings. Harmful algal blooms can be directly toxic to aquatic organisms, resulting in impairment of populations and even local species extinctions (Burkholder 1998). Algal blooms known as brown or red tides are intense growths of algal species which release neurotoxins into the water column. The neurotoxins can harm some forms of aquatic life. These types of blooms usually necessitate beach closings and temporary bans on seafood harvesting from the water body because consuming shellfish from affected waters can cause shellfish poisoning in humans (Burkholder 1998).

Algal blooms caused by excessive nutrient input into a water body can significantly alter aquatic community structure. Increased algal growth can reduce available light for submerged aquatic vegetation, reducing photosynthesis levels and decreasing food and habitat for benthic organisms. Decomposition of excessive algal biomass reduces dissolved oxygen levels in surface waters, lessening the amount available to other aquatic organisms and either impairing their functioning or suffocating them.

Other types of algal blooms may not be as directly harmful to humans but can still cause economic damage because the increased levels of organic material associated with the blooms may clog drinking water intakes or cause an unpleasant taste in drinking water (Boyd 1990).

3.2.2 Polycyclic Aromatic Hydrocarbons

Because PAHs are not specified as a type of 305(b) impairment in the EPA WATERS database, surface waters in the United States are not listed as impaired by this specific pollutant. However, PAHs are a known carcinogen in many animals and are suspected to be carcinogenic in humans as well, though no

specific information on concentrations of concern is currently available (USEPA 2008e). PAHs tend to travel attached to sediment, settling with it on water body beds. They may then be ingested by benthic dwelling or feeding organisms, facilitating PAH transfer up the food chain (USEPA 2004b). PAHs also have bioaccumulation potential and have been found to cause birth defects and liver and blood problems in some animals (USEPA 2008e). PAHs in benthic sediments have been specifically linked to liver abnormalities in bottom-dwelling fish species (Malins et al. 1988; Vethaak and Rheinhallt 1992).

3.2.3 Metals

Metals as a group (excluding mercury, which is considered separately) are considered an impairment category in the EPA WATERS database. As with sediment and nutrient listings, it should be noted that the 305(b) data only assess 19 percent of rivers and streams; 37 percent of lakes, ponds, and reservoirs; and 35 percent of bays and estuaries, so the actual number of waters impaired by nutrient pollution is likely to be much higher than the 2002 305(b) data indicate.

Table 3-3 details 305(b) metal impairment listings by EPA region. More than 33,000 miles of assessed rivers and streams were impaired by metals, along with nearly 1.3 million acres of lakes, ponds, and reservoirs and 4,700 square miles of bays and estuaries. Region 1 shows the greatest length of assessed rivers and streams with metal impairment at 11,606, while Region 6 accounts for the largest share of lake, pond, and reservoir metal impairment. As with nutrient impairment, Region 3 shows the largest area of bays and estuaries impaired by metals, though Region 6 also reported nearly 1,700 square miles of metal-impaired bays and estuaries in 2002. WATERS data on current 303(d) lists show that metals account for 12.5 percent of all 303(d) waters awaiting TMDL development.

Table 3-3: Metal Impairment in 305(b)-Assessed Waters[1]

EPA Region	River, Stream (miles)		Lake[2], Pond, Reservoir (acres)		Bay, Estuary (sq. miles)	
	Assessed	Impaired	Assessed	Impaired	Assessed	Impaired
1	21,699.5	11,606.2	1,491,595.5	398,797.8	1,018.7	21.3
2	12,814.8	0.0	409,319.9	0.0	1,675.9	0.0
3	127,468.7	5,284.2	271,042.5	30,168.4	4,996.1	2,996.5
4	66,507.3	517.2	1,173,482.6	32,155.0	329.8	0.0
5	128,144.0	6,824.4	3,699,833.3	55,137.0	86.0	0.0
6	49,969.2	3,376.0	2,726,913.2	634,326.8	7,802.7	1,689.4
7	56,467.7	2,151.1	693,980.7	135,478.0	0.0	0.0
8	110,169.4	1,178.3	1,912,016.9	1,681.6	0.0	0.0
9	41,477.6	293.0	1,272,016.2	169.0	958.8	7.8
10	80,414.5	2,324.6	1,181,681.6	0.0	13,577.7	12.0
Total	**695,132.8**	**33,555.0**	**14,831,882.3**	**1,287,913.6**	**30,445.7**	**4,727.0**

1 Values are preliminary pending correction of EPA's publically available 305(b) dataset

2 Does not include Great Lakes

Source: WATERS (USEPA 2008g)

Metals tend to adsorb to sediment in the aquatic environment. Metals can therefore accumulate on water body beds as they settle out of the water column with sediments. Metals deposited in the substrate may be consumed by benthic organisms and then progress through the food chain, potentially harming larger organisms. Trace amounts of metals are necessary for normal biological functions in many aquatic species. However, concentrations above the trace level can be acutely or chronically toxic, adversely affecting behavior and reproductive success in both aquatic species and waterfowl. Aquatic ecosystem

exposure to metals has also been shown to reduce photosynthetic efficiency and algal colonization, lowering primary production rates and diminishing food available to other organisms.

When present at sufficiently high levels, metals can cause dredged sediment to be classified as toxic, making its disposal more costly. 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.

4 Overview of Benefits from Regulation

This chapter provides an overview of the potential benefits to society related to the reduced sediment discharges from construction sites that will result from the proposed regulation. Sediments and other pollutants from construction sites 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.

4.1 Conceptual Framework for Valuation of Environmental Services

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

$$Q = f(S(R), P, A) \quad \text{(Eq. 4-1)}$$

For this analysis, water quality changes are estimated in terms of sediment concentrations in receiving waters using SPARROW (see *Chapter 6* and *Appendix A* 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, sediment discharges is 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 sediment discharges 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 sediment 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; USEPA 2004b). *Chapter 10, Section 10.1.1,* 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. 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:

$$X = f(Q, I) \quad \text{(Eq. 4-2)}$$

The vector of environmental services or human uses *(X)* affected by changes in water quality and sediment discharges could include (van Houtven et al. 2005):

- Services received through purchases of market goods *(Z)* produced with surface water resources (e.g., commercial fish)
- Goods, services, and activities produced within the household *(H),* such as recreation days
- Public sector services *(G)* such as drinking water quality
- Existence values *(N)* 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]

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

The relationship between individual or household utility *(U)* and water quality *(Q)* can be more formally described though the following utility function (van Houtven et al. 2005):

$$U = U\,(Z\,(Q),\ H\,(Q, I),\ G\,(Q),\ N\,(Q),\ Y\,;\ B) \qquad \text{(Eq. 4-3)}$$

Individuals are assumed to derive utility from *Z, H, G,* and *N,* as well as from other goods and services *(Y),* which are not related to water resources. A subset of household goods and services *(H),* such as recreation, are "produced" with water quality *(Q)* and other inputs *(I)* 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 Eq. 4-3 with respect to an income *(M)* constraint produces the following indirect utility function:

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

Where *P* represents the vector of prices associated with goods and services *(Z, H, G, 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 surplus *(CS)* measure from the following equation:

$$V(Q_0, P_0, Y_0; B) = V(Q_1, P_0, Y_0 - CS; B) \qquad \text{(Eq. 4-5)}$$

Other things being equal, an individual is better off with a higher level of water quality (Q_1) than a lower level (Q_0). CS 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.

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

$$CS = CS(Q_0, Q_1, P_0, Y_0; B) \qquad \text{(Eq. 4-6)}$$

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. 2005).

4.2 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.

4.2.1 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 latter 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).

Maintenance of Navigational Waterways

The primary effect of sediment discharges on navigation and transportation is accumulation of sediment in navigational channels and harbors (USEPA 2004b). Keeping these areas passable requires extensive dredging, which can be costly. Additionally, since dredged material may contain significant quantities of toxics and heavy metals, dredging can create new environmental problems (Clark et al. 1985). 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. *Chapter 7* of this report details methods and presents results of EPA's analysis of benefits to navigation expected from the regulation.

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. To replace this capacity, sediment must be dredged from reservoirs, or new reservoirs must be constructed (Clark et al. 1985). Sediment can also affect evaporation rates at reservoirs. 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 could be positive or negative. 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. *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 2007a). 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 proposed regulation. *Chapter 9* of this report details methods and presents results of EPA's analysis of benefits to drinking water treatment plants. The presence of pollutants or dissolved minerals in drinking water may also affect the flavor and odor of water. Although EPA did not estimate a value of improved taste and odor of drinking water, this benefit is 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).

- *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. However, at least one positive effect is possible as well. Since turbidity may reduce the rate at which water bodies absorb solar heat, more turbid water bodies may supply cooler water, which in turn could improve the efficiency of cooling water systems (Clark et al. 1985). The proposed 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. 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, resulting in inhibited plant growth and reduced crop value and marketability. Furthermore, irrigation water that contains dissolved salts or pollutants can harm crops and damage soil quality (Clark et al. 1985). The proposed 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.

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. *Chapter 2* of this report provides some detail on aquatic life impacts from increased sediment concentrations in fresh and marine waters. Reducing sediment 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 is 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).

Public and Private Property Ownership

- *Property Values.* Aesthetic degradation of land and water resources resulting from sediment 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 has 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 1993). Stabilization of stream banks leads to an increase in the value of surrounding property (Streiner 1996). Therefore, the proposed regulation is expected to enhance nearby property value in two ways: (1) by improving 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) is 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). Clark et al. (1985) estimated flooding damages attributable to sediment discharges to be $1.5 billion (2008$), annually. 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 1985). The proposed 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.

4.2.2 Nonmarket Benefits

EPA expects that this regulation will reduce the sediment discharges to surface waters and will enhance or protect aquatic ecosystems currently under stress. The decrease in sediment loading 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. Improvements in water quality such as decrease in turbidity will also favor increased recreational activities 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.

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). Recreation is a typical example of nonmarket direct use of water resources. Improved water quality from reducing sediment 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.

Adverse impacts of sediment 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, Section 2.3.2, Aquatic Life Impacts of Sediment,* 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 pollutions reduce the aesthetics of the water body, which may reduce the anglers' utility of their fishing experience. Additionally, sediment 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 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. At $7 million a life, the value of preventing these fatalities is $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 associated with sediment discharges 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.

The proposed regulation is 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.

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.

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 discharges from construction sites and the specific ecological changes, from a lack of water quality monitoring data for many locations, and from time lags between water quality changes and changes in species population and composition. In addition, it is 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; U.S. 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 (Johnston et al. 2005; USEPA 2000b). 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 B* as well as in sources such as Johnston et al. (2005; 2006), Bateman and Jones (2003), Shrestha et al. (2007), Rosenberger and Phipps (2007), and USEPA (2004c). *Chapter 10* of this report describes application of the meta-analysis results for estimating benefits of water quality improvements resulting from the regulation.

4.3 Summary of Effects

Ultimately, changes in sedimentation related to the proposed regulation affect humans through influences on the production of goods and services that enter into individuals' utility functions. *Table 4-1* summarizes some of the important ways in which sediment discharges from construction sites affects 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), quantify but not monetize other benefits (changes in TSS concentrations in coastal waters), 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 4-1: Summary of Benefits from Reducing Sediment 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, 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

5 Pollutant Load Modeling

To estimate the environmental changes that would occur as a result of the regulation, EPA modeled construction site pollutant discharges under baseline conditions and under each of several regulatory scenarios. EPA used the Revised Universal Soil Loss Equation (RUSLE, pronounced "Russell") (Renard 1997) to estimate sediment generated from a series of model construction sites. By evaluating the particle size distribution of soils present in areas surrounding 11 indicator cities, EPA estimated the fraction of sediment discharges that would be removed under baseline conditions and under a series of regulatory options. EPA scaled up the model site results to the state level based on the amount of construction activity occurring in each state. EPA then allocated state-level loads to RF1 watersheds within the state based on the percentage of development occurring in each RF1 watershed.

The procedures that EPA used in the RUSLE analysis and in the pollutant load reductions estimates are described in Chapter 10 of EPA's *Development Document for Proposed Effluent Guidelines and Standards for the Construction and Development Category* (USEPA 2008b). The results of this analysis are also detailed in that document.

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

- Option 1 would establish minimum sizing criteria for sediment basins used at construction sites with 10 or more disturbed acres draining to one location. Under this option, permittees would be required to install sediment basins that either provide 3,600 cubic feet per acre of runoff storage, or are designed to store runoff from the local two-year, 24-hour storm event. This option also includes requirements for implementing a variety or erosion and sediment controls 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 acres would be required to meet a numeric turbidity limit in stormwater discharges from the site. The numeric turbidity standard would be applicable to stormwater discharges for all storm events up to the local two-year, 24-hour event. The turbidity standard would only apply to construction sites located in areas where the rainfall runoff erosivity factor (R-factor) as defined in RUSLE is greater than or equal to 50 and if the soils on the site contain 10 percent or more by mass of soil particles smaller than 2 microns in diameter.
- 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. The turbidity standard would apply to all sites, regardless of soil types or R-factor. The turbidity standard would apply to all stormwater discharges for all storm events up to the local two-year, 24-hour event.

Table 5-1 presents a national summary of sediment loadings calculated by EPA for the baseline and three policy options, as well as reductions under each policy option. Option 1 is expected to result in a

reduction of 1.2 percent of construction sediment discharges nationally, while Option 2 and Option 3 are expected to result in reductions of 46.0 percent and 87.8 percent, respectively. Total reductions under Option 2 are 13.21 million tons of sediment, and 25.21 million tons under Option 3, compared to just 340,000 tons under Option 1.

Table 5-1: Baseline Construction Sediment Loading Summary and Post-Compliance Reductions

Scenario	Sediment Loading (million tons)	Sediment Reduction (million tons)	% Reduction from Baseline
Baseline	28.70	-	-
Option 1	28.36	0.34	1.2%
Option 2	15.49	13.21	46.0%
Option 3	3.49	25.21	87.8%

Source: USEPA analysis

Table 5-2, *Table 5-3*, and *Table 5-4* present estimates of the amount of sediment entering reaches from construction related activity under each of these three options, by EPA region. Each table shows the baseline amount of sediment entering reaches as well as post-compliance conditions and reductions for a regulatory option. Under Option 1, Region 4 shows the largest absolute and percentage reduction in sediment loadings, at 2.63 million tons and 2.3 percent, respectively. Reductions in other regions are 1 percent or less of baseline sediment loadings with the exception of Region 8, which shows a 1.5 percent reduction. Option 2 shows more significant reductions in all regions, with Region 4 again showing the largest reduction in terms of absolute mass of sediment prevented from entering surface waters (4.85 million tons), though Region 7 shows a larger percentage reduction, at nearly 60 percent of construction sediment discharges prevented from entering waters. Option 3 reduces construction site sediment discharges by 87 to 88 percent in all regions, with Region 4 showing the largest absolute reduction of 11.46 million tons of construction sediment discharges prevented from entering surface waters.

Table 5-2: Baseline Construction Sediment Loading Summary and Option 1 Reductions by EPA Region

EPA Region	Baseline Sediment Loading (million tons)	Option 1		
		Sediment Loading (million tons)	Sediment Reduction (million tons)	% Reduction from Baseline
1	0.11	0.11	0.00	0.5%
2	0.52	0.51	0.01	1.0%
3	2.63	2.63	0.00	0.0%
4	13.01	12.72	0.30	2.3%
5	2.78	2.76	0.02	0.9%
6	3.71	3.71	0.00	0.0%
7	4.64	4.63	0.00	0.1%
8	0.35	0.35	0.01	1.5%
9	0.11	0.11	0.00	0.0%
10	0.83	0.83	0.00	0.0%
National Total	**28.70**	**28.36**	**0.34**	**1.2%**

Source: USEPA analysis

Table 5-3: Baseline Construction Sediment Loading Summary and Option 2 Reductions by EPA Region

EPA Region	Baseline Sediment Loading (million tons)	Option 2		
		Sediment Loading (million tons)	Sediment Reduction (million tons)	% Reduction from Baseline
1	0.11	0.08	0.03	29.2%
2	0.52	0.27	0.25	47.6%
3	2.63	1.06	1.57	59.6%
4	13.01	8.16	4.85	37.3%
5	2.78	1.39	1.39	50.0%
6	3.71	1.66	2.06	55.4%
7	4.64	1.87	2.77	59.8%
8	0.35	0.27	0.08	22.3%
9	0.11	0.10	0.01	7.3%
10	0.83	0.62	0.21	25.2%
National Total	**28.70**	**15.49**	**13.21**	**46.0%**

Source: USEPA analysis

Table 5-4: Baseline Construction Sediment Loading Summary and Option 3 Reductions by EPA Region

EPA Regions	Baseline Sediment Loading (million tons)	Option 3		
		Sediment Loading (million tons)	Sediment Reduction (million tons)	% Reduction from Baseline
1	0.11	0.01	0.09	87.8%
2	0.52	0.06	0.46	87.9%
3	2.63	0.32	2.31	88.0%
4	13.01	1.56	11.46	88.0%
5	2.78	0.35	2.43	87.5%
6	3.71	0.47	3.24	87.3%
7	4.64	0.57	4.07	87.7%
8	0.35	0.04	0.31	88.0%
9	0.11	0.01	0.09	88.0%
10	0.83	0.10	0.73	88.3%
National Total	**28.70**	**3.49**	**25.21**	**87.8%**

Source: USEPA analysis

6 Water Quality Modeling

6.1 SPARROW Model Documentation[4]

6.1.1 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 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 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 B* of this report provides more detail on the nature of the general SPARROW methodology. *Appendix C* provides more detail on the suspended sediment SPARROW model developed by USGS (Schwarz 2008a).

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

[4] Discussion adapted from Schwarz et al., USGS (2006).

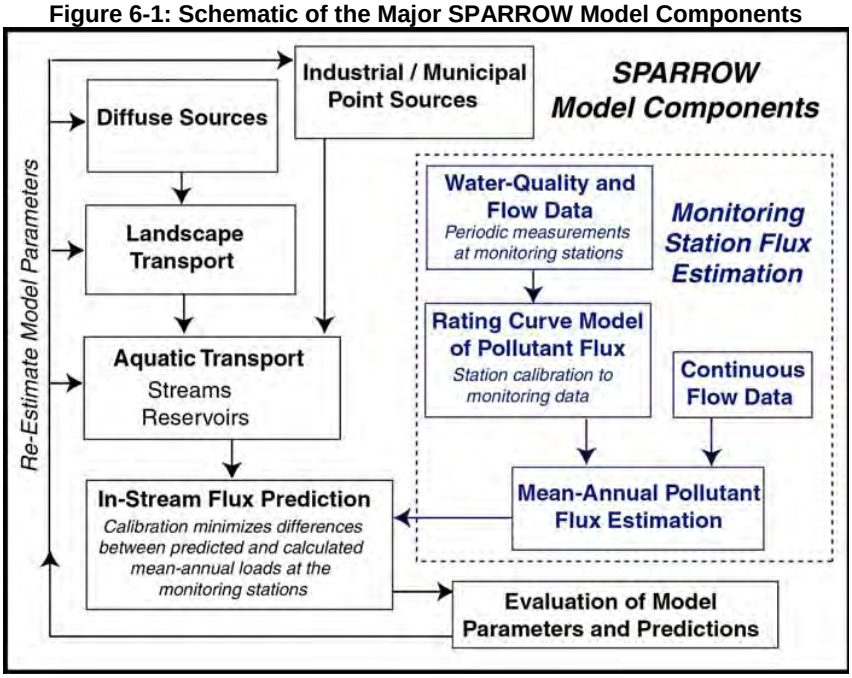
Practices (BMP) or regulatory approaches. For the analysis described in this document, SPARROW is used to evaluate the effectiveness of a range of regulatory options for controlling and mitigating sediments associated with construction activities, with explicit focus on the impacts of such options on in-stream water quality as measured by Total Suspended Solids (TSS). Determining regulatory option impacts on national water quality through physical experimentation rather than through modeling would be extremely difficult. 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.

SPARROW models are expressed in the form of a mass balance. Because the dependent variable in SPARROW models (the mass of contaminant that passes a specific stream location per unit time, also referred to herein as loading) is linearly related to all sources of contaminant mass in the model, all accounting rules relating to the conservation of mass apply. Mass accounting in SPARROW models is also 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 loading accounting, whereby loading 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 loading 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. Loading 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

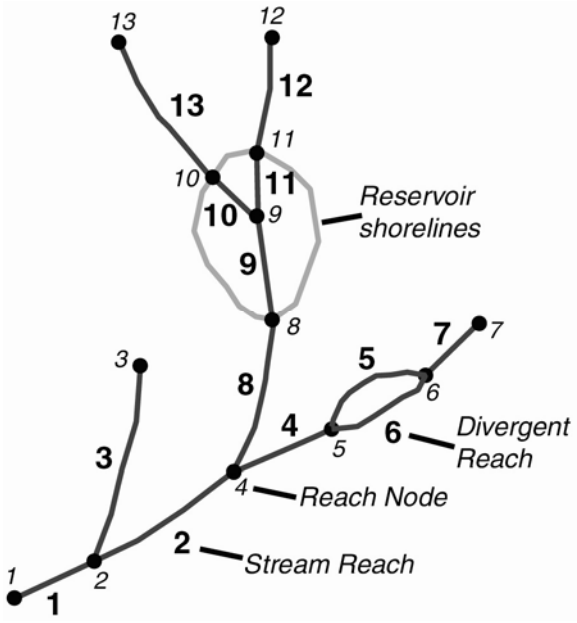


Source: Modified from Alexander, Elliott, et al. (2002)

The estimation of a SPARROW model requires estimates of long-term mean loading 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 loading 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 loading 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 loadings 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 load or loading 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 loading from the stream network may occur at points where flow is diverted, and

loading 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. For loading 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. 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 loadings. 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 mass loadings generated by a particular type of a contaminant 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 loading 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 water body.

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 loading. 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 preliminary 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 C* of this document. The analysis is based on loading 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, and 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 loading. Per unit area, Federal nonforested 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 the EPA (1996) 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

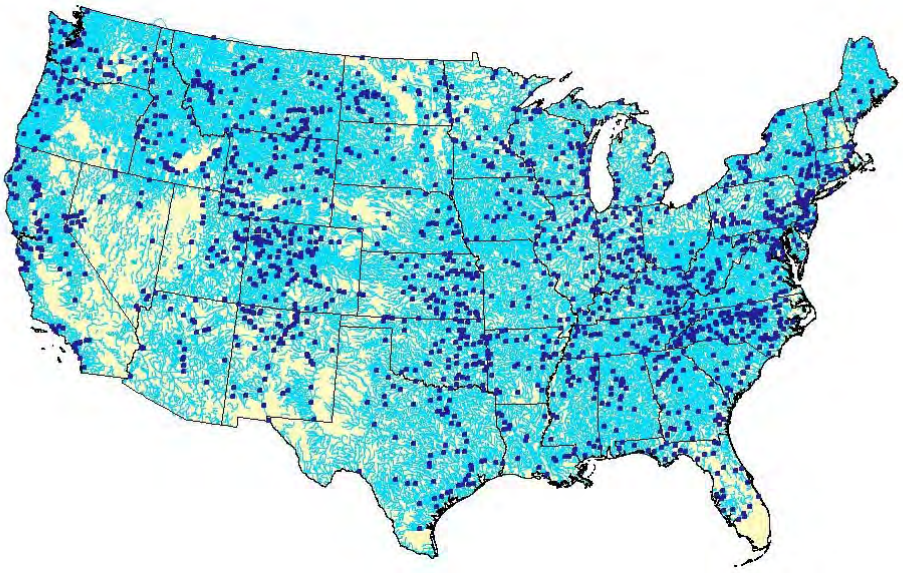
[5] Adapted from Schwarz (2008a).

hydraulic load information for approximately 2,000 additional large reservoirs from the National Inventory of Dams (NID, USACE 1996). The dependent variable in the SPARROW sediment model is given by long-term mean sediment loading, 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 government. 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 (Smiley 2007). 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 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 Preliminary 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 preliminary 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 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 indicates medium sized streams (flow 500-1,000 cfs) have a statistically significant negative rate of in-stream attenuation, indicating that medium streams are a source of sediment, in addition to the stream channel source identified above. The negative rate of in-stream attenuation implies that sediment loading is proportionally enhanced as it passes through medium sized streams. This enhancement process is inconsistent with mass balance and is yet to be explained. In-stream 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.

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 loading 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 loading 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 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 in the SPARROW model of an explicit term for construction loading requires 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 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 loading 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 Water Quality Modeling Results

This section presents the results of water quality modeling using the SPARROW model for sediment. SPARROW 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 72 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. *Figure 6-4* illustrates the highly uneven distribution of construction acres among RF1 watersheds during the 1992-2001 time period. *Figure 6-5* shows the cumulative distribution of acres of construction by RF1 watersheds in decreasing order of construction acres by watershed. 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 4 percent of RF1 watersheds containing the largest number of construction acres per watershed encompassed, as a group, almost 3 million construction acres, or 56 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)

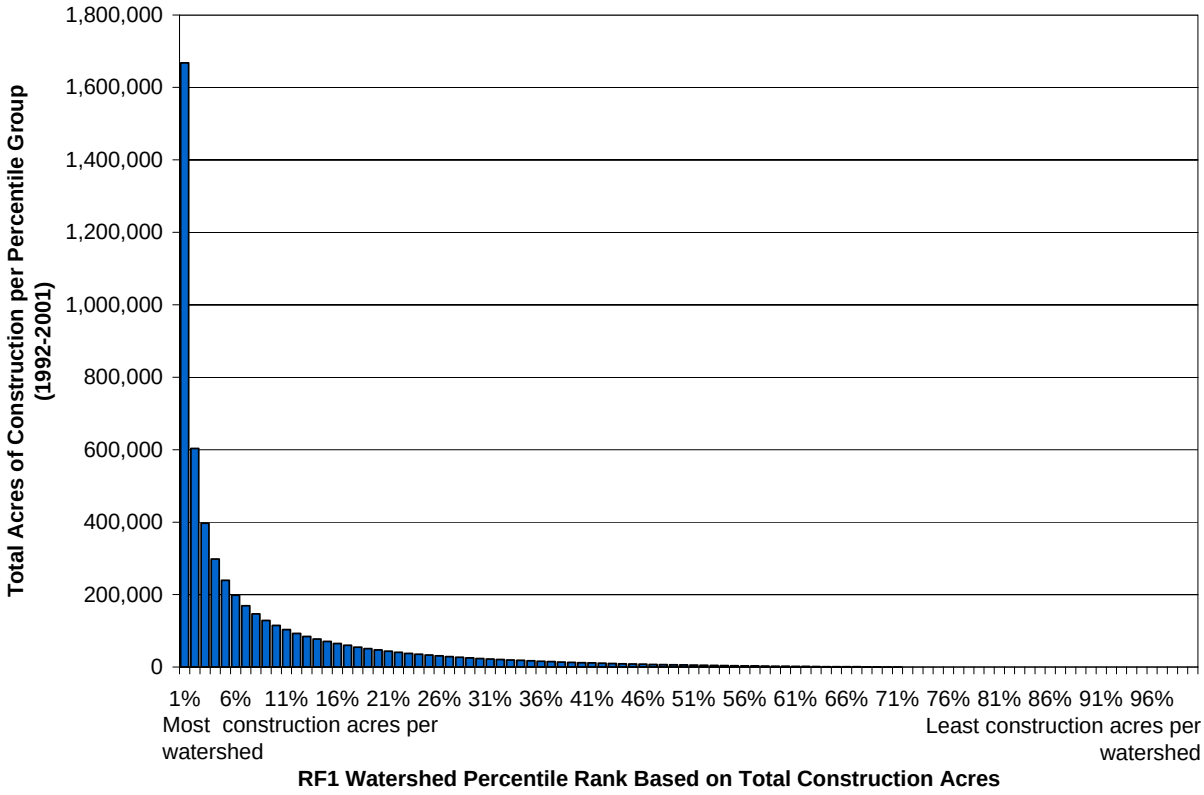
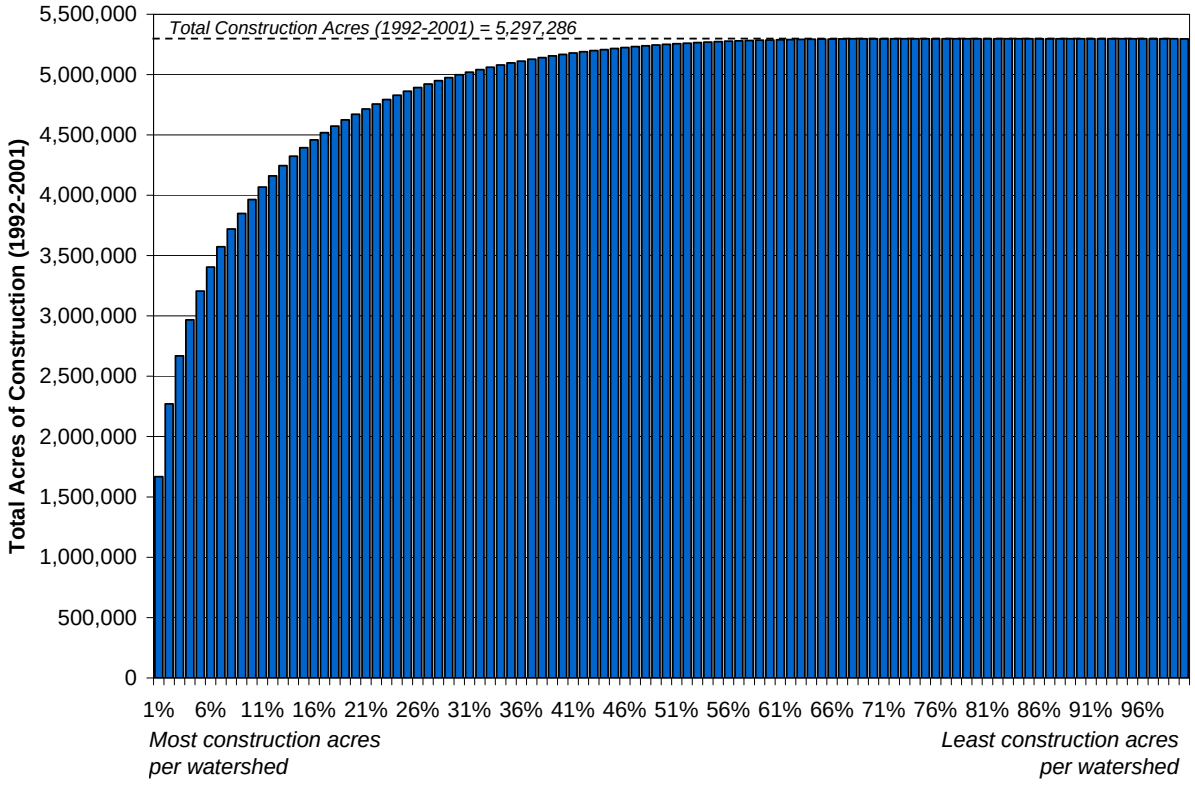


Figure 6-5: Cumulative Plot of Construction Acres by RF1 Watershed (1992-2001)



Because construction sites are distributed so unevenly among watersheds, this chapter summarizes water quality information in two ways. The first 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. The 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 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 in the tables below, however, are not intended to reflect 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 a finer level of time resolution.

EPA expects that, in general, surface water TSS concentrations in the short time periods following storm events would be higher than the concentrations presented in the tables below. Sediment concentration reductions associated with the three proposed regulatory options would be higher, as well. 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.

A second important factor to consider when examining the data in the tables below is that construction site sediment 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 concentrations among different groups of reaches, as reflected by reaches having progressively greater construction acreage within their watershed (presented in *Table 6-2* through *Table 6-6*) and among the various EPA Regions (presented in *Table 6-4*, *Table 6-5*, and *Table 6-6*). 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.

Despite the limited detail they provide on temporal and geospatial variations in TSS concentration, the summaries are nevertheless useful for interpreting broad-scale changes in water quality as a result of the proposed regulatory options.

When summarizing TSS concentrations, EPA calculated statistics based on reach-length weighted TSS concentrations to account for the wide variation among reaches, as noted in the tables. EPA believes that this adjustment provides a more representative characterization of the distribution of TSS concentration by giving more weight to the longer reaches. Additionally, EPA has generally excluded from the distribution TSS concentrations estimated on reaches that have baseline TSS concentrations above the 95th percentile (6,049.7 mg/L), since these reaches could potentially be considered outliers.

6.3.1 Estimated Changes in TSS Concentration in Watersheds with Most Developed Acres (1992-2001)

Table 6-2 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 nearly three-quarters of the increase in developed land in the coterminous United States, while the top 25 percent account for nearly all developed land (91.8 percent). These watersheds are also distributed unevenly across EPA regions both in terms of total river miles and the number of water body reaches, as shown in *Table 6-3*. For example, approximately 22 percent of river miles within the top 1 percent are found in Region 4 even though the region accounts for approximately 15 percent of overall RF1 river miles.

These differences are reflected in TSS concentration estimates shown in *Table 6-4*, *Table 6-5*, and *Table 6-6* for each of the three top groups, respectively, and across the three regulatory options (see *Chapter 5* 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 three 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 6-2: Top Percent of Reaches by Increase in Developed Land Area (1992-2001)

Percent of RF1 Watersheds	Number of RF1 Watersheds[1]	% 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	612	31.5%	1.7	2,727.5	94.5	5.0%
Top 10% of RF1 Watersheds	6,112	74.9%	4.0	648.5	484.8	25.7%
Top 25% of RF1 Watersheds	15,310	91.8%	4.9	317.6	870.4	46.2%
Coterminous U.S. (100% of Reaches)	61,121	100.0%	5.3	86.6	1,885.3	100.0%

[1] RF1 watersheds included in count are those for which both NLCD and SPARROW data are available.
Top 1% of RF1 watersheds had 1,304 acres or greater of developed land area (1992-2001); Top 10% of RF1 watersheds had 178 acres or greater of developed land area (1992-2001); Top 25% of watersheds had 52 acres or greater of developed land area (1992-2001).

Table 6-3: Distribution of Top RF1 Watersheds by Increase in Developed Land Area (1992-2001)

EPA Region	Top 1% of Watersheds		Top 10% of Watersheds		Top 25% of Watersheds		Coterminous U.S. (100% of Reaches)	
	River Miles	% of Miles	River Miles	% of Miles	River Miles	% of Miles	River Miles	% of Miles
1	146	1%	1,872	2%	5,178	2%	18,303	3%
2	122	1%	2,834	2%	6,714	3%	16,099	2%
3	917	7%	9,757	8%	19,453	8%	33,601	5%
4	3,058	22%	27,552	24%	57,298	23%	94,458	15%
5	2,419	17%	16,761	15%	34,112	14%	71,510	11%
6	2,999	22%	24,913	22%	46,585	19%	98,642	15%
7	1,400	10%	12,412	11%	26,276	11%	60,887	9%
8	853	6%	9,535	8%	26,285	11%	130,190	20%
9	1,643	12%	6,072	5%	12,191	5%	56,463	9%
10	342	2%	3,860	3%	11,061	5%	69,398	11%
TOTAL	**13,897**	**100%**	**115,568**	**100%**	**245,153**	**100%**	**649,552**	**100%**

Table 6-4: Estimated Changes in RF1 Reach TSS Concentration by Policy Option: Top 1% of RF1 Watersheds by Increase in Developed Land Area (1992-2001)

Compliance Scenario	Median TSS (mg/L) [1]	River-Mile Weighted Average TSS (mg/L) [1]	Total Improved		Reduction in TSS Concentrations (mg/L)									
					0< TSS <1		1≤ TSS <5		5≤ TSS <10		10≤ TSS <50		50≤ TSS	
			RF1 Miles	% of Total 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	RF1 Miles	% of Total Improved Miles
Baseline	350.59	703.72	-	-	-	-	-	-	-	-	-	-	-	-
Option 1	350.59	703.07	4,314	31.01%	3,532	25.39%	425	3.06%	111	0.80%	220	1.58%	25	0.18%
Option 2	337.59	680.48	12,619	90.71%	4,176	30.02%	3,480	25.02%	1,025	7.37%	2,536	18.23%	1,403	10.09%
Option 3	296.41	660.78	13,031	93.67%	2,007	14.43%	3,941	28.33%	1,720	12.36%	2,665	19.16%	2,698	19.39%

*Total miles (13,911) are based on top 1% of reaches by area of urban development from 1992 to 2001.

[1] River-mile weighted.

Table 6-5: Estimated Changes in RF1 Reach TSS Concentration by Policy Option: Top 10% of RF1 Watersheds by Increase in Developed Land Area (1992-2001)

Compliance Scenario	Median TSS (mg/L) [1]	River-Mile Weighted Average TSS (mg/L) [1]	Total Improved		Reduction in TSS Concentrations (mg/L)									
					0< TSS <1		1≤ TSS <5		5≤ TSS <10		10≤ TSS <50		50≤ TSS	
			RF1 Miles	% of Total 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	RF1 Miles	% of Total Improved Miles
Baseline	248.34	642.70	-	-	-	-	-	-	-	-	-	-	-	-
Option 1	248.05	642.43	32,455	28.08%	29,358	25.40%	2,242	1.94%	395	0.34%	365	0.32%	96	0.08%
Option 2	239.16	631.25	106,876	92.46%	42,851	37.07%	30,838	26.68%	11,396	9.86%	17,267	14.94%	4,523	3.91%
Option 3	231.65	622.91	109,397	94.64%	27,264	23.59%	23,495	20.33%	33,818	29.26%	15,977	13.82%	8,843	7.65%

*Total miles (115,588) are based on top 10% of reaches by area of urban development from 1992 to 2001.

[1] River-mile weighted.

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

Compliance Scenario	Median TSS (mg/L) [1]	River-Mile Weighted Average TSS (mg/L) [1]	Total Improved		Reduction in TSS Concentrations (mg/L)									
					0< TSS <1		1≤ TSS <5		5≤ TSS <10		10≤ TSS <50		50≤ TSS	
			RF1 Miles	% of Total 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	RF1 Miles	% of Total Improved Miles
Baseline	248.39	632.84	-	-	-	-	-	-	-	-	-	-	-	-
Option 1	248.34	632.68	69,845	28.49%	65,058	26.54%	3,714	1.51%	536	0.22%	441	0.18%	96	0.04%
Option 2	241.80	625.02	225,465	91.97%	104,586	42.66%	65,838	26.86%	21,668	8.84%	27,205	11.10%	6,169	2.52%
Option 3	236.29	619.46	231,932	94.61%	74,942	30.57%	75,640	30.85%	29,476	12.02%	39,725	16.20%	12,149	4.96%

*Total miles (245,153) are based on top 25% of reaches by area of urban development from 1992 to 2001.

[1] River-mile weighted.

6.3.2 Estimated Changes in TSS Concentrations for all RF1 Watersheds Receiving Sediment Loading from Construction Sites

Table 6-7 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 three regulatory options (see *Chapter 5* for option descriptions). Information is presented for all reaches in the RF1 network that receive sediment loading from construction sites.

Table 6-7 also presents information on the distribution of TSS concentrations under the baseline scenario representing current conditions and the three 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 25.11 and 3,035.28 mg/L respectively. Median (50th percentile) TSS concentrations are approximately 261.49 mg/L under the baseline scenario. Under Option 1, the median concentration decreases 0.17 mg/L to 261.32 mg/L. Median concentrations under Option 2 decrease 4.88 mg/L to 256.61 mg/L; under Option 3, they decrease 7.72 mg/L to 253.77 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. TSS concentrations in *Table 6-7* representing the 5th, 25th, 50th, 75th, and 95th percentiles provide information on the distribution of TSS concentration at the national level.

Table 6-7: TSS Concentration Distribution Based on SPARROW Output for 43,821 RF1 Reaches Receiving Construction Sediment Discharges

Policy Option	Reach Count	River Miles	Average TSS (mg/L)[1, 2]	Distribution of TSS Concentrations[1, 2]				
				5th Percentile	25th Percentile	50th Percentile (Median)	75th Percentile	95th Percentile
Baseline	43,821	517,982	678.20	25.11	96.51	261.49	723.68	3,035.28
Option 1	43,821	517,982	678.11	25.04	96.43	261.32	723.60	3,035.28
Option 2	43,821	517,982	673.69	23.47	94.18	256.61	715.05	3,029.70
Option 3	43,821	517,982	670.41	20.92	91.68	253.77	711.39	3,019.24

[1] Based on reach-length weighted distribution

[2] Excludes potential outliers (i.e., reaches with baseline TSS concentration above the 95th percentile)

Source: USEPA analysis

Table 6-8 through *Table 6-11* detail TSS concentrations by EPA region under baseline conditions and for each of the three 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 900 mg/L under all scenarios. Median TSS concentrations in these regions—ranging from 333 to 919 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 185 mg/L. Under Option 1, TSS concentrations decline a small amount (by 0.3 percent) from baseline concentrations in Region 4 and are nearly identical in all other regions. Under Options 2 and 3, reductions in TSS concentrations are greater and are present in all regions.

Table 6-8: Summary of Baseline TSS Concentration in RF1 Reaches Receiving Construction Sediment Discharges, by EPA Region

EPA Region	Reach Count	River Miles	Average TSS (mg/L)[1,2]	Distribution of TSS Concentrations[1]				
				5th Percentile	25th Percentile	50th Percentile (Median)	75th Percentile	95th Percentile
1	1,863	15,599	76.61	6.19	25.46	40.19	58.56	143.52
2	1,467	15,439	109.44	10.26	33.80	75.49	134.93	305.92
3	2,673	32,676	188.39	30.16	73.23	135.12	225.28	484.68
4	8,870	91,882	229.24	27.32	86.95	174.16	307.20	592.39
5	5,170	68,476	297.41	16.54	58.88	185.23	425.17	890.25
6	5,774	83,009	1,161.98	64.68	233.85	522.55	1,602.60	4,376.59
7	4,019	56,489	1,208.77	161.91	443.79	918.55	1,662.43	3,372.74
8	6,931	81,993	1,326.23	63.42	271.95	782.73	1,931.92	4,438.09
9	2,471	34,393	947.23	28.07	140.02	332.81	1,237.99	4,113.00
10	4,583	38,026	236.80	19.41	52.92	108.85	245.39	880.86
National	**43,821**	**517,982**	**678.20**	**25.11**	**96.51**	**261.49**	**723.68**	**3,035.28**

[1] Based on reach-length weighted distribution

[2] Excludes potential outliers (i.e., reaches with baseline TSS concentration above the 95th percentile)

Source: USEPA analysis

Table 6-9: Summary of Option 1 TSS Concentration in RF1 Reaches Receiving Construction Sediment Discharge, by EPA Region

EPA Region	Reach Count	River Miles	Average TSS (mg/L)[1, 2]	Distribution of TSS Concentrations[1]				
				5th Percentile	25th Percentile	50th Percentile (Median)	75th Percentile	95th Percentile
1	1,863	15,599	76.61	6.19	25.46	40.19	58.56	143.52
2	1,467	15,439	109.43	10.26	33.80	75.49	134.93	305.91
3	2,673	32,676	188.39	30.16	73.23	135.09	225.28	484.68
4	8,870	91,882	228.79	27.30	86.83	173.56	306.10	590.91
5	5,170	68,476	297.40	16.54	58.88	185.23	425.17	890.24
6	5,774	83,009	1,161.98	64.68	233.85	522.55	1,602.60	4,376.59
7	4,019	56,489	1,208.76	161.91	443.79	918.55	1,662.43	3,372.72
8	6,931	81,993	1,326.22	63.42	271.93	782.73	1,931.79	4,437.80
9	2,471	34,393	947.23	28.07	140.02	332.81	1,237.99	4,113.00
10	4,583	38,026	236.80	19.41	52.92	108.85	245.39	880.86
National	**43,821**	**517,982**	**678.11**	**25.04**	**96.43**	**261.32**	**723.60**	**3,035.28**

[1] Based on reach-length weighted distribution

[2] Excludes potential outliers (i.e., reaches with baseline TSS concentration above the 95th percentile)

Source: USEPA analysis

Table 6-10: Summary of Option 2 TSS Concentration in RF1 Reaches Receiving Construction Sediment Discharge, by EPA Region

EPA Region	Reach Count	River Miles	Average TSS (mg/L)[1, 2, 3]	Distribution of TSS Concentrations[1]				
				5th Percentile	25th Percentile	50th Percentile (Median)	75th Percentile	95th Percentile
1	1,863	15,599	76.33	6.13	25.17	39.75	58.25	143.36
2	1,467	15,439	108.97	10.25	33.75	74.89	133.98	305.14
3	2,673	32,676	184.63	28.68	71.75	132.22	222.62	475.38
4	8,870	91,882	222.60	26.77	84.59	167.76	300.46	574.48
5	5,170	68,476	296.52	16.20	58.23	185.18	422.97	889.99
6	5,774	83,009	1,152.75	54.06	224.94	512.77	1,593.86	4,376.36
7	4,019	56,489	1,199.25	158.86	434.18	913.59	1,650.71	3,284.40
8	6,931	81,993	1,325.85	63.42	271.89	782.73	1,931.36	4,436.97
9	2,471	34,393	946.91	28.07	140.01	332.81	1,237.99	4,112.90
10	4,583	38,026	231.62	12.48	47.92	105.14	242.08	867.20
National	**43,821**	**517,982**	**673.69**	**23.47**	**94.18**	**256.61**	**715.05**	**3,029.70**

[1] Based on reach-length weighted distribution.
[2] Average excludes potential outliers (values above the 95th percentile).
Source: USEPA analysis

Table 6-11: Summary of Option 3 TSS Concentration in RF1 Reaches Receiving Construction Sediment Discharge, by EPA Region

EPA Region	Reach Count	River Miles	Average TSS (mg/L)[1, 2, 3]	Distribution of TSS Concentrations[1]				
				5th Percentile	25th Percentile	50th Percentile (Median)	75th Percentile	95th Percentile
1	1,863	15,599	76.33	5.87	25.02	39.62	58.14	142.02
2	1,467	15,439	108.97	9.98	33.57	74.31	133.89	304.46
3	2,673	32,676	184.63	27.56	71.28	130.96	221.41	466.99
4	8,870	91,882	222.60	19.93	80.71	163.53	293.93	560.85
5	5,170	68,476	296.52	15.74	57.32	184.78	421.09	888.25
6	5,774	83,009	1,152.75	49.93	221.93	511.88	1,590.63	4,374.28
7	4,019	56,489	1,199.25	158.60	427.47	909.96	1,645.77	3,284.26
8	6,931	81,993	1,325.85	62.97	268.12	779.09	1,929.57	4,431.54
9	2,471	34,393	946.91	28.07	139.55	332.35	1,225.99	4,112.12
10	4,583	38,026	231.62	0.00	42.00	100.99	237.22	860.97
National	**43,821**	**517,982**	**670.41**	**20.92**	**91.68**	**253.77**	**711.39**	**3,019.24**

[1] Based on reach-length weighted distribution.
[2] Excludes potential outliers (i.e., reaches with baseline TSS concentration above the 95th percentile).
Source: USEPA analysis

The following tables describe the number of RF1 reach miles improved and the distribution of TSS concentration reductions for each of three options analyzed for this regulation. As shown in *Table 6-12*, which summarizes the effects of sediment reduction at the national level, Option 3 improves 94.3 percent of modeled RF1 river miles receiving construction sediment loading. Option 2 is predicted to lead to improvements in 90.9 percent of RF1 river miles, and Option 1 is predicted to improve 29.9 percent of RF1 river miles. Under Option 3, approximately half of the improved river miles show reductions in TSS concentrations of less than one mg/L, and another 26.9 percent show reductions in TSS concentrations

between 1 and 5 mg/L. Nearly 3 percent of improved RF1 river miles are expected to see larger reductions in TSS concentrations of 50 mg/L or more. Under Option 2, 62.6 percent of improved RF1 river miles experience TSS reductions of less than 1 mg/L, and 7.1 percent are improved by a reduction in TSS concentrations greater than 10 mg/L. Of the nearly 30 percent of RF1 river miles improved under Option 1, 96.4 percent show improvements of less than 1 mg/L.

Table 6-13 summarizes Option 1's sediment reductions at the regional level. Nationwide, 96.4 percent of improved RF1 river miles under Option 1 experience TSS concentration reductions less than 1 mg/L. Out of ten regions, only EPA Region 4 is estimated to experience larger improvements in surface water quality, with 8.6 percent of improved RF1 river miles projected to have TSS concentrations reduced between 1 and 5 mg/L and 2.5 percent of river miles expected to have TSS concentrations reduced by 5 mg/L or greater.

As shown in *Table 6-14*, Option 2 is estimated to reduce TSS concentrations by greater than 50 mg/L in 7,014 river miles nationwide that receive construction sediment loading, and 33,577 river miles by 10 to 50 mg/L. Smaller improvements of 1 to 5 mg/L are predicted in 106,250 RF1 river miles, and 294,802 river miles are expected to experience modest improvements in TSS concentrations (less than 1 mg/L). EPA Region 4 is estimated to benefit the most in terms of total improved river miles (91,296 miles). Regions 6 and 7 are expected to see a greater percentage of reaches with reductions in TSS greater than 10 mg/L. Regions 1, 2, 5, 8, and 9 are predicted to have more than 75 percent of TSS concentration reductions be less than 1 mg/L.

Option 3 (*Table 6-15*) is estimated to reduce TSS concentrations by more than 50 mg/L in 14,106 RF1 miles receiving construction sediment loading and by between 10 and 50 mg/L in 50,845 miles, meaning that more than 13 percent of improvements under this option are TSS reductions of 10 mg/L or greater. Region 4 has the largest gain in terms of number of river miles (91,297 miles) improved, though Region 7 is expected to have a greater proportion of improved reaches with larger reductions in sediment concentration, with nearly 28 percent of improved RF1 miles having 10 mg/L or greater reductions in TSS concentrations. Regions 1, 2, 5, 8, 9, and 10 have smaller reductions in most of their improved reaches though these regions do have more reaches with TSS reductions 1 mg/L or greater than they do under Options 1 and 2.

Table 6-12: Summary of Estimated Changes in TSS Concentration by Policy Option

Compliance Scenario	Reduction in TSS Concentrations (mg/L)										Total Improvements	
	0< Δ TSS <1		1≤ Δ TSS <5		5≤ Δ TSS <10		10≤ Δ TSS <50		50≤ Δ TSS			
	RF1 Miles	% of Improved Miles	RF1 Miles	% of Improved Miles	RF1 Miles	% of Improved Miles	RF1 Miles	% of Improved Miles	RF1 Miles	% of Improved Miles	RF1 Miles	% of Total RF1 Miles[1]
Option 1	149,029	96.36%	4,388	2.84%	639	0.41%	506	0.33%	96	0.06%	154,658	29.86%
Option 2	294,803	62.63%	106,250	22.57%	29,085	6.18%	33,578	7.13%	7,014	1.49%	470,730	90.88%
Option 3	249,489	51.10%	131,368	26.91%	42,416	8.69%	50,844	10.41%	14,105	2.89%	488,224	94.26%

[1] Calculated as a percentage of the RF1 river miles that receive construction sediment loading (517,982 miles in total), as modeled by SPARROW.

Source: USEPA analysis

Table 6-13: Water Quality Impacts: Improvements in TSS Concentrations from Baseline to Option 1

EPA Region	Total Improved River Miles	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	
		River Miles	% of Total Improved Miles	River Miles	% of Total Improved Miles	River Miles	% of Total Improved Miles	River Miles	% of Total Improved Miles	River Miles	% of Total Improved Miles
1	3,165	3,165	100.00%	0	0.00%	0	0.00%	0	0.00%	0	0.00%
2	3,048	3,048	100.00%	0	0.00%	0	0.00%	0	0.00%	0	0.00%
3	917	917	100.00%	0	0.00%	0	0.00%	0	0.00%	0	0.00%
4	48,589	43,187	88.88%	4,170	8.58%	630	1.30%	506	1.04%	96	0.20%
5	32,874	32,846	99.91%	28	0.09%	0	0.00%	0	0.00%	0	0.00%
6	3,836	3,836	100.00%	0	0.00%	0	0.00%	0	0.00%	0	0.00%
7	12,581	12,458	99.02%	123	0.98%	0	0.00%	0	0.00%	0	0.00%
8	46,256	46,171	99.82%	67	0.14%	9	0.02%	9	0.02%	0	0.00%
9	505	505	100.00%	0	0.00%	0	0.00%	0	0.00%	0	0.00%
10	2,896	2,896	100.00%	0	0.00%	0	0.00%	0	0.00%	0	0.00%
National Total	**154,667**	**149,029**	**96.35%**	**4,388**	**2.84%**	**639**	**0.41%**	**515**	**0.33%**	**96**	**0.06%**

Source: USEPA analysis

Table 6-14: Water Quality Impacts: Improvements in TSS Concentrations from Baseline to Option 2

EPA Region	Total Improved River Miles	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	
		River Miles	% of Total Improved Miles	River Miles	% of Total Improved Miles	River Miles	% of Total Improved Miles	River Miles	% of Total Improved Miles	River Miles	% of Total Improved Miles
1	15,391	14,659	95.24%	604	3.92%	105	0.68%	23	0.15%	0	0.00%
2	15,367	13,760	89.54%	1,412	9.19%	191	1.24%	4	0.03%	0	0.00%
3	32,308	16,846	52.14%	11,777	36.45%	1,859	5.75%	1,665	5.15%	161	0.50%
4	91,296	38,783	42.48%	32,405	35.49%	8,785	9.62%	9,278	10.16%	2,045	2.24%
5	68,136	56,203	82.49%	9,786	14.36%	1,269	1.86%	811	1.19%	67	0.10%
6	75,721	27,413	36.20%	25,861	34.15%	9,273	12.25%	11,010	14.54%	2,164	2.86%
7	49,840	19,574	39.27%	14,590	29.27%	5,451	10.94%	8,519	17.09%	1,706	3.42%
8	64,596	58,576	90.68%	4,903	7.59%	718	1.11%	399	0.62%	0	0.00%
9	27,026	26,231	97.06%	696	2.58%	14	0.05%	49	0.18%	36	0.13%
10	31,048	22,757	73.30%	4,216	13.58%	1,421	4.58%	1,819	5.86%	835	2.69%
National Total	**470,729**	**294,802**	**62.63%**	**106,250**	**22.57%**	**29,086**	**6.18%**	**33,577**	**7.13%**	**7,014**	**1.49%**

Source: USEPA analysis

Table 6-15: Water Quality Impacts: Improvements in TSS Concentrations from Baseline to Option 3

EPA Region	Total Improved River Miles	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	
		River Miles	% of Total Improved Miles	River Miles	% of Total Improved Miles	River Miles	% of Total Improved Miles	River Miles	% of Total Improved Miles	River Miles	% of Total Improved Miles
1	15,392	13,572	88.18%	1,465	9.52%	193	1.25%	138	0.90%	24	0.16%
2	15,367	12,165	79.16%	2,783	18.11%	334	2.17%	85	0.55%	0	0.00%
3	32,309	13,046	40.38%	13,882	42.97%	2,573	7.96%	2,362	7.31%	446	1.38%
4	91,297	22,405	24.54%	36,548	40.03%	12,637	13.84%	15,192	16.64%	4,515	4.95%
5	68,135	49,131	72.11%	14,972	21.97%	2,286	3.36%	1,515	2.22%	231	0.34%
6	75,720	19,929	26.32%	23,368	30.86%	13,413	17.71%	15,050	19.88%	3,960	5.23%
7	49,839	15,142	30.38%	15,542	31.18%	5,382	10.80%	10,719	21.51%	3,054	6.13%
8	71,666	53,098	74.09%	14,387	20.08%	2,651	3.70%	1,376	1.92%	154	0.21%
9	30,746	24,774	80.58%	4,133	13.44%	908	2.95%	831	2.70%	100	0.33%
10	37,755	26,226	69.46%	4,290	11.36%	2,040	5.40%	3,577	9.47%	1,622	4.30%
National Total	**488,226**	**249,488**	**51.10%**	**131,370**	**26.91%**	**42,417**	**8.69%**	**50,845**	**10.41%**	**14,106**	**2.89%**

Source: USEPA analysis

6.3.3 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 449 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 8*. The policy options reduce this amount to various degrees. As shown in *Table 6-16*, Option 1 reduces the amount of sediment accumulating in reservoirs slightly as compared to current conditions (by 123,600 cubic yards or approximately 0.03 percent). Most of this reduction is concentrated in Region 4. Options 2 and 3 show much greater reductions in reservoir sedimentation, with estimated reductions of 5.9 and 9.8 million cubic yards, respectively (or 1.3 and 2.2 percent). They also show more geographically distributed effects, with significant reductions observed in Regions 6, 4, 7, and 10.

Table 6-16: Sediment Accumulation in Reservoirs by Policy Option and EPA Region[1]

EPA Region	Sediment Accumulation (thousand cubic yards)				Reduction in Sediment Accumulation (thousand cubic yards)		
	Baseline	Option 1	Option 2	Option 3	Option 1	Option 2	Option 3
1	1,318	1,318	1,308	1,298	0.1	9.6	20.3
2	2,769	2,769	2,758	2,749	0.1	11.6	20.4
3	5,457	5,457	5,378	5,342	0.2	78.9	115.4
4	46,353	46,233	44,426	43,012	120.6	1,927.7	3,341.0
5	36,094	36,093	35,940	35,854	1.4	154.1	240.4
6	130,271	130,271	127,468	126,000	0.2	2,802.6	4,270.5
7	42,126	42,126	41,412	41,129	0.3	714.0	997.8
8	127,509	127,509	127,500	127,444	0.7	9.0	65.7
9	42,583	42,583	42,579	42,537	0.1	3.6	45.2
10	14,918	14,918	14,732	14,195	0.0	185.8	722.9
Total	**449,399**	**449,275**	**443,502**	**439,559**	**123.7**	**5,896.9**	**9,839.6**

[1] Excludes reservoirs located on "outlier" reaches (i.e., reaches with baseline TSS concentration above the 95th percentile).

6.3.4 Hypothetical "No Discharge" Scenario

Table 6-17 presents SPARROW output for the baseline scenario representing current conditions and a hypothetical scenario in which all sediment discharges from construction activity are prevented from entering surface waters. SPARROW estimates that approximately 767 million cubic yards of sediment are transported in RF1 reaches each year, and 450 million cubic yards settle in reservoirs. SPARROW estimates that removing all construction site sediment contributions would result in a 3.48 percent reduction in national sediment loadings in RF1 reaches and a 2.49 percent reduction in national reservoir sediment attenuation. 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. Therefore, while construction sediment discharges represent approximately 3.48 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 (~0.03% of coterminous United States), construction sites have a disproportionately high rate of sediment contribution to surface waters relative to other land uses.

Table 6-17: Total Yearly Sediment from All Sources for Baseline and Zero Discharge Scenarios Based on SPARROW Output for 62,370 RF1 Reaches

Scenario	River Miles	In-Stream Sediment Flux (millions of cubic yards per year)[1,2]	% Reduction	Reservoir Sediment Attenuation (millions of cubic yards per year)[1,3]	% Reduction	Total Sediment Flux (millions of cubic yards per year)[1]	% Reduction
Baseline	650,043	317.51	-	449.78	-	767.29	-
Zero Discharges	650,043	302.01	4.88%	438.56	2.49%	740.57	3.48%

[1] Based on all reaches estimated by SPARROW (62,370 RF1 Reaches). One cubic yard = 1,127 kg.
[2] In-stream sediment flux estimated by subtracting total sediment flux minus total reservoir sediment attenuation.
[3] Total includes reaches that do not receive construction sediment loading. Values are therefore slightly greater than reported in other tables that include only reaches that receive construction sediment loading (449.40 million cubic yards for the baseline scenario).

Source: USEPA analysis

6.4 Extension of SPARROW to Estuaries and Coastal Waters

EPA used SPARROW output to evaluate sediment deliveries to estuaries, as well. Estimates of ambient concentrations of sediment [Total Suspended Solids (TSS)] within estuaries were calculated using the Dissolved Concentration Potential (DCP) approach (Versar, Inc./USEPA, 1997). The DCP approach was developed by the National Oceanic and Atmospheric Administration (NOAA) to predict the ambient concentration of conserved contaminants that are introduced into an estuary and are then subject to mixing and dilution. An advantage to this approach is that published values of DCP have been provided by NOAA for most major estuaries in the coterminous United States. Using this approach, EPA was able to estimate TSS concentrations associated with annual quantities of sediment discharged into each estuary (as predicted by SPARROW) without detailed numerical simulation modeling of estuaries. The DCP approach presumes 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 met 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 approximately applicable for estimating changes in estuary water quality associated with changes in construction site sediment discharges, and the approach may be capable of generating credible results for most major estuaries.

NOAA has calculated DCP values for most major estuaries in the Southeast (NOAA and USEPA, 1989a); Northeast (NOAA and USEPA, 1989b); Gulf of Mexico (NOAA and USEPA, 1989c); and West Coast (NOAA and USEPA, 1991) of the coterminous United States. Calculated DCP values for 22 West Coast, 24 Northeast, 23 Gulf, and 17 Southeastern U.S. estuaries are presented in tabular form in USEPA (1997) and in *Table 6-18* to *Table 6-21*. 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 and shoreline reaches draining to estuaries were aggregated on the basis of 5-digit NOAA Estuarine Drainage Area (EDA) codes. Estimated loadings are 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 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 Suspended Sediment Concentration (SSC) and Total Suspended Solids (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 6.3: Water Quality Modeling Results.* 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 complicated, 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 varied from below 1 mg/L (Northeastern estuaries) to a maximum of 830 mg/L (San Antonio Bay, TX), and averaged 71 mg/L over all estuaries and all scenarios. Estimates of TSS as calculated using DCP appear in *Table 6-18* for Southeastern Estuaries, *Table 6-19* for Gulf of Mexico Estuaries, *Table 6-20* for Northeastern Estuaries, and *Table 6-21* for West Coast Estuaries. A number of estimated TSS values fall outside of anticipated ranges, and the sources of variation across estuaries require careful investigation. Negative concentrations under Option 3 for Indian River, FL, Tampa Bay, FL, Puget Sound, WA and Yaquina Bay, OR represent artifacts of SPARROW estimation and are not to be interpreted physically.

Average reductions in estuarine TSS concentrations relative to baseline values were less than 1 percent for Option 1; approximately 3.3 percent for Option 2; and 8.7 percent for Option 3. Considerable regional variation was observed. In Northeastern estuaries, very little change in TSS concentrations were observed across management options. This primarily reflects the relatively low baseline TSS concentrations,

averaging only 13 mg/L across Northeastern estuaries. By contrast, baseline TSS concentrations across all estuaries averaged 71 mg/L, and TSS concentrations in Gulf of Mexico estuaries averaged 149 mg/L. The largest regional changes observed, a 13 percent average decrease relative to baseline TSS concentrations for Option 3, were found in the West Coast estuaries.

A preliminary examination was made of SPARROW-simulated TSS concentrations relative 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, SPARROW/DCP estimates of TSS for Delaware Bay (40.33 mg/L) are very close to observed values (38.45), and virtually identical for Long Island Sound (9.75 predicted vs. 9.8 observed). Results for San Francisco Bay illustrate the difficulty in comparing model predictions with observations. SPARROW/DCP TSS estimates of 90 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). These and related results suggest that SPARROW loading predictions, transformed by DCP assumptions to estuarine ambient concentrations (TSS), 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. Results are summarized in *Table 6-22* for seven major estuaries.

Table 6-18: TSS Concentrations (mg/L) Estimated by DCP, Southeastern Estuaries

Estuary	DCP	Base TSS (mg/L)	Option 1 TSS (mg/L)	% change rel. to base	Option 2 TSS (mg/L)	% change rel. to base	Option 3 TSS (mg/L)	% change rel. to base
Albemarle/Pamlico Sound, NC, VA	0.14	34.3	34.2	-0.1%	33.9	-1.0%	33.7	-1.7%
Altamaha River, GA	0.37	82.8	82.3	-0.6%	78.6	-5.1%	74.7	-9.7%
Biscayne Bay, FL	0.4	0.2	0.2	0.0%	0.2	0.0%	0.2	-4.7%
Bogue Sound, NC	1.47	3.0	3.0	-0.1%	3.0	-0.8%	2.9	-1.4%
Broad River, SC	4.92	16.3	16.3	0.0%	15.9	-2.7%	14.5	-10.9%
Cape Fear River, NC	0.61	69.3	69.2	-0.1%	68.7	-0.9%	68.3	-1.5%
Charleston Harbor, SC	0.44	2.6	2.6	0.0%	2.6	-2.3%	2.4	-9.2%
Indian River, FL	1.02	5.3	5.3	0.0%	5.2	-1.1%	N/A	N/A
N and S Santee Rivers, SC	2.91	456.5	456.2	-0.1%	450.5	-1.3%	441.1	-3.4%
New River, NC	7.68	19.0	19.0	-0.1%	18.9	-0.5%	18.9	-0.9%
Ossabaw Sound, GA	1.99	81.0	80.9	-0.2%	79.9	-1.4%	78.9	-2.7%
Savannah River, SC, GA	0.43	61.1	61.0	-0.2%	58.9	-3.7%	54.4	-11.0%
St. Catherines/Sapelo Sound, GA	7.56	17.9	17.8	-0.7%	16.8	-6.2%	15.8	-11.9%
St. Helena Sound, SC	0.95	10.8	10.8	0.0%	10.8	-0.8%	10.5	-3.3%
St. Johns River, FL	0.83	13.5	13.5	0.0%	13.5	-0.3%	7.4	-45.4%
St. Andrews/St. Simons Sound, GA	2.43	47.4	47.3	-0.3%	46.0	-2.9%	44.7	-5.7%
Winyah Bay, SC, NC	0.39	108.6	108.5	-0.1%	107.4	-1.1%	106.4	-2.0%

Source: USEPA analysis

Table 6-19: TSS Concentrations (mg/L) Estimated by DCP, Gulf of Mexico Estuaries

Estuary	DCP	Base TSS (mg/L)	Option 1 TSS (mg/L)	% change rel. to base	Option 2 TSS (mg/L)	% change rel. to base	Option 3 TSS (mg/L)	% change rel. to base
Apalachee Bay, FL, GA	0.36	8.2	8.2	0.0%	8.2	-0.5%	4.7	-43.0%
Apalachicola Bay, FL	0.17	27.7	27.1	-2.0%	22.7	-18.2%	18.0	-35.1%
Arkansas Bay, TX	6.02	138.6	138.6	0.0%	138.3	-0.3%	138.1	-0.4%
Atchafalaya and Vermillion Bays, LA	0.04	251.0	251.0	0.0%	248.0	-1.2%	246.7	-1.7%
Brazos River, TX	1.11	595.6	595.6	0.0%	593.6	-0.3%	592.5	-0.5%
Calcasieu Lake, LA	1.18	48.3	48.3	0.0%	47.5	-1.7%	47.0	-2.9%
Charlotte Harbor, FL	0.76	15.5	15.5	0.0%	15.5	-0.1%	13.5	-13.0%
Choctawhatchee Bay, FL, AL	0.7	51.6	51.5	-0.1%	51.2	-0.6%	50.5	-2.1%
Corpus Cristi Inner Harbor, TX	4.67	554.9	554.9	0.0%	549.1	-1.0%	546.0	-1.6%
Galveston Bay, TX	0.4	93.3	93.3	0.0%	85.1	-8.7%	80.8	-13.4%
Laguna Madre, TX	0.34	24.0	24.0	0.0%	24.0	-0.3%	23.9	-0.4%
Matagorda Bay, TX	0.81	186.5	186.5	0.0%	185.4	-0.6%	184.7	-1.0%
Mississippi River, LA, MS	0.01	194.8	194.8	0.0%	193.3	-0.8%	192.7	-1.1%
Mississippi Sound, MS, LA, AL	0.17	128.0	128.0	0.0%	119.6	-6.5%	115.5	-9.8%
Mobile Bay, AL	0.08	86.9	86.7	-0.2%	85.2	-2.0%	83.9	-3.4%
Pensacola Bay, FL	0.46	43.9	43.9	-0.1%	43.6	-0.7%	43.0	-2.1%
Perdido Bay, FL, AL	2.89	35.6	35.5	-0.1%	35.2	-1.1%	34.4	-3.4%
Sabine Lake, LA, TX	0.38	76.3	76.3	0.0%	67.0	-12.3%	61.9	-18.9%
San Antonio Bay, TX	1.3	829.4	829.4	0.0%	817.0	-1.5%	810.4	-2.3%
St. Andrew Bay, FL	0.76	2.3	2.3	0.0%	2.3	-0.2%	1.7	-23.9%
Suwanee River, FL	0.38	24.6	24.6	0.0%	24.5	-0.4%	23.7	-3.4%
Tampa Bay, FL	1.03	10.5	10.5	0.0%	10.3	-1.8%	N/A	N/A
Ten Thousand Islands, FL	1.94	1.7	1.7	0.0%	1.7	0.0%	1.6	-2.4%

Source: USEPA analysis

Table 6-20: TSS Concentrations (mg/L) Estimated by DCP, Northeastern Estuaries

Estuary	DCP	Base TSS (mg/L)	Option 1 TSS (mg/L)	% change rel. to base	Option 2 TSS (mg/L)	% change rel. to base	Option 3 TSS (mg/L)	% change rel. to base
Barnegat Bay, NJ	1.36	3.7	3.7	0.0%	3.7	-0.2%	3.7	-0.7%
Blue Hill Bay, ME	1.03	1.7	1.7	0.0%	1.7	-2.0%	1.6	-3.8%
Buzzards Bay, MA	1.04	3.8	3.8	0.0%	3.8	-0.1%	3.7	-1.2%
Cape Cod Bay, MA	0.69	0.5	0.5	0.0%	0.5	-0.1%	0.5	-1.4%
Casco Bay, ME	0.61	3.3	3.3	0.0%	3.3	-0.6%	3.3	-1.1%
Chesapeake Bay, VA, MD, DE, PA, DC	0.072	92.9	92.9	0.0%	91.4	-1.6%	90.6	-2.5%
Chincoteague Bay, MD, VA	3.08	2.4	2.4	0.0%	2.4	-0.9%	2.4	-1.5%
Delaware Bay, DE, NJ, PA, MD	0.14	40.3	40.3	0.0%	39.9	-1.2%	39.7	-1.7%
Englishman Bay, ME	0.92	4.7	4.7	0.0%	4.7	-0.1%	4.7	-0.2%
Gardiners Bay, NY	1.77	2.0	2.0	0.0%	2.0	-0.2%	2.0	-0.3%
Great Bay, NE, NH	1.54	7.6	7.6	0.0%	7.6	-0.4%	7.5	-1.4%
Great South Bay, NY	5.07	12.0	12.0	0.0%	12.0	-0.2%	12.0	-0.4%
Hudson River/Raritan Bay, NY, NJ, MA, CT	0.2	52.4	52.4	0.0%	52.1	-0.7%	51.8	-1.3%
Long Island Sound, NY, CT, MA	0.054	9.7	9.7	0.0%	9.7	-0.2%	9.7	-0.5%
Massachusetts Bay, MA	0.27	2.4	2.4	0.0%	2.4	-0.2%	2.3	-2.2%
Merrimack River, NH., MA	1.01	32.0	32.0	0.0%	31.9	-0.1%	31.7	-0.8%
Muscongus Bay, ME	2.25	6.8	6.8	0.0%	6.6	-3.8%	6.3	-7.4%
Narragansett Bay, MA, RI	0.52	8.4	8.4	0.0%	8.4	-0.1%	8.3	-0.9%
Narragauges Bay, ME	1.54	4.3	4.3	0.0%	4.3	-0.3%	4.3	-0.5%
Passaquoddy Bay, ME	0.27	1.5	1.5	0.0%	1.5	-0.1%	1.5	-0.1%
Penobscot Bay, ME	0.13	7.1	7.1	0.0%	7.1	-0.7%	7.0	-1.3%
Saco Bay, ME, NH	0.45	4.2	4.2	0.0%	4.2	-0.3%	4.2	-0.8%
Sheepscop Bay, ME, NH	0.088	0.2	0.2	0.0%	0.2	-1.7%	0.2	-3.3%

Source: USEPA analysis

Table 6-21: TSS Concentrations (mg/L) Estimated by DCP, West Coast Estuaries

Estuary	DCP	Base TSS (mg/L)	Option 1 TSS (mg/L)	% change rel. to base	Option 2 TSS (mg/L)	% change rel. to base	Option 3 TSS (mg/L)	% change rel. to base
Alsea River, OR	2.086	37.4	37.4	0.0%	36.5	-2.2%	35.0	-6.4%
Columbia River, WA, OR	0.032	76.0	76.0	0.0%	75.0	-1.3%	72.7	-4.3%
Coos Bay, OR	2.27	51.3	51.3	0.0%	45.5	-11.4%	34.7	-32.4%
Eel River, CA	0.39	65.3	65.3	0.0%	65.3	0.0%	65.1	-0.3%
Grays Harbor, WA	0.278	21.2	21.2	0.0%	18.7	-11.6%	11.6	-45.2%
Humboldt Bay, CA	8	38.7	38.7	0.0%	38.6	-0.2%	37.9	-1.9%
Klamath River, OR, CA	0.31	79.7	79.7	0.0%	79.6	-0.2%	78.5	-1.6%
Monterey Bay, CA	1.05	148.4	148.4	0.0%	148.4	0.0%	148.3	0.0%
Nehalam River, OR	1.556	38.5	38.5	0.0%	37.2	-3.4%	34.8	-9.5%
Netarts Bay, OR	12.564	15.7	15.7	0.0%	14.6	-7.4%	12.4	-21.0%
Puget Sound, WA	0.039	9.7	9.7	0.0%	6.9	-28.4%	N/A	N/A
Rogue River, OR	0.556	125.2	125.2	0.0%	124.9	-0.2%	124.3	-0.7%
San Diego Bay, CA	12.31	88.9	88.9	0.0%	88.9	0.0%	88.9	-0.1%
San Francisco Bay, CA	0.104	90.4	90.4	0.0%	90.4	0.0%	90.2	-0.3%
San Pedro Bay, CA	4.41	104.4	104.4	0.0%	104.4	0.0%	104.3	-0.1%
Santa Monica Bay, CA	1.371	40.9	40.9	0.0%	40.9	0.0%	40.9	0.0%
Siletz Bay, OR	2.424	42.8	42.8	0.0%	39.5	-7.9%	33.2	-22.5%
Siuslaw River, OR	1.853	46.5	46.5	0.0%	45.6	-1.9%	44.0	-5.4%
Tillamook Bay, OR	1.049	22.1	22.1	0.0%	20.7	-6.3%	18.1	-17.9%
Umpqua River, OR	0.793	109.5	109.5	0.0%	109.0	-0.5%	108.0	-1.4%
Willapa Bay, WA	0.466	12.1	12.1	0.0%	9.4	-22.3%	1.5	-87.2%
Yaquina Bay, OR	3.356	25.7	25.7	0.0%	9.3	-63.8%	N/A	N/A

Source: USEPA analysis

Table 6-22: Comparison between SPARROW-DCP Estimated and Observed Suspended-Sediment Concentrations (mg/L)

Estuary	DCP	Area (km^2)	SPARROW baseline	Data mean	Data std. dev.	Coefficient variation	Data count	Data source
Chesapeake Bay	0.072	168,794	92.92	13.82	7.88	0.57	94	b
Columbia River	0.032	617,376	76.02	8.33	13.36	1.60	132	c
Delaware Bay	0.14	33,777	40.33	38.45	32.01	0.83	90	a
Hudson River	0.2	41,793	52.44	16.07	22.97	1.43	40	a
Long Island Sound	0.054	42,222	9.75	9.80	11.18	1.14	176	a
Puget Sound	0.039	29,500	9.68	4.31	1.36	0.32	60	d
San Francisco Bay (1)	0.104	116,772	90.39	132.13	470.58	3.56	71	e
San Francisco Bay (2)	0.104	116,772	90.39	33.88	46.16	1.36	5176	f

Sources:
a. EMAP VA Province.
b. EMAP Mid-Atlantic, VA Province.
c. EMAP West Coast/Oregon DEQ.
d. EMAP West Coast/Washington Dep. Env.
e. EMAP West Coast/NOAA.
f. USGS Water Quality of San Francisco Bay.

6.5 Use of APEX-SWAT to Evaluate Construction Sediment Regulatory Options: Case Study, Dallas, TX

6.5.1 Overview

The micro-watershed model APEX (**A**gricultural **P**olicy – **E**nvironmental e**X**tender; Williams et al. 2008), in conjunction with the watershed-scale model SWAT (**S**oil and **W**ater **A**ssessment **T**ool; Arnold et al. 1998), is being used to evaluate a range of regulatory options for mitigating sediment associated with construction activities. The APEX-SWAT study is configured to complement the nationwide SPARROW model evaluation described in *Sections 6.1–6.3* of this document. The studies are complementary in several respects: APEX-SWAT employs a physically based simulation modeling approach to the evaluation of regulatory options, whereas SPARROW employs what is primarily a statistically estimated nonlinear regression approach. The SPARROW study is national in scope, at the spatial resolution of the Reach File 1 (RF1) stream network, while the APEX-SWAT approach involves a case study of a single urban area – Dallas, Texas on the Trinity River – with spatial resolution at plot/multi-field to sub-catchment scale. Finally, SPARROW is specified to predict the long-term annual or seasonal mean contaminant loadings (long-term annual loading associated with specific land use and management practices); and APEX-SWAT examines transient conditions, with respect to both land disturbance and to weather and flow conditions.

The APEX-SWAT study is motivated by several insights emerging from the national SPARROW sediment modeling study. The fundamental spatial unit within the national SPARROW model is the RF1 stream reach and associated catchment. Even in rapidly urbanizing regions, sediment loadings associated with construction activities, while perhaps the dominant sediment sources within the specific tributary reaches containing active construction sites, may contribute only modestly to observed sediment loadings as measured or simulated at the RF1 level, depending on the magnitude of water and sediment contributions from other tributaries and sediment sources within the RF1 catchment. Spatial averaging of loadings at the RF1 level may obscure the degree to which significant changes (and hence potential benefits) occur within specific construction-affected reaches in response to respective sediment abatement practices.

As a related issue, while SPARROW is in principle able to account for the influences of the position of the sediment source (construction site) within the landscape via statistical evaluation of land-to-water delivery processes over wide areas, in practice the sensitivity of such insights is limited by the spatial resolution of model landscape units. In the national SPARROW model, this resolution is 1 km^2. By contrast, APEX-SWAT simulation can be conducted over a range of spatial resolutions down to 1 hectare or finer, if desired, and details of the overland flowpath, such as length, slope, hydraulic roughness, and vegetative cover, can be simulated explicitly. Finally, APEX-SWAT simulation can be used to examine the influences of rainfall intensity and duration, overlaid on variations in antecedent soil moisture and seasonally determined vegetative cover conditions, on soil erosion and sediment mobilization associated with a specific sediment abatement practice. Via continuous simulation, decades of daily (and sub-daily) weather variability can be evaluated.

The sections below provide additional details on the APEX-SWAT analysis. Results from the analysis are not available at this time. EPA plans to present the results from this analysis in a future version of this document.

6.5.2 Objectives

The primary objective of the APEX-SWAT model case study is to use continuous, dynamic simulation to evaluate the respective water quality impacts of a range of sediment abatement options designed to reduce or to eliminate the delivery to surface waters of sediments and associated contaminants originating on construction sites. Related objectives are (1) to evaluate the influence of the construction site setting within the landscape on loadings to the surface flow system; (2) to evaluate the influence of meteorological characteristics of the study setting on sediment loadings; and (3) to assess the spatial variation in sediment loadings at tributary scale within the larger watershed. Attributes of interest with respect to landscape setting include proximity of the construction site to the nearest channel; characteristics of the construction site (e.g., size, shape, slope) and of the intervening flowpath, including length, slope, hydraulic roughness, and vegetative cover; and the impacts of buffers, filters or related structures on sediment loadings. Attributes of interest with respect to meteorological conditions include rainfall intensity, duration, inter-storm duration and antecedent soil moisture conditions. The intent is to examine the combined influences of sediment abatement practices themselves, landscape attributes, and meteorological conditions on sediment loadings, in order to acquire a more comprehensive understanding of the likely water quality impacts of proposed sediment abatement options under heterogeneous conditions.

6.5.3 APEX-SWAT

The farm- and field-scale model APEX (Williams et al. 2008) has been combined with the watershed simulation model SWAT (Arnold et al. 1998) to provide a modeling framework within which detailed evaluations of soil conservation, tillage, and nutrient management practices and their impacts on soil, nutrient, and contaminant runoff at field scale can be integrated to provide estimates of water quality impacts at watershed scale. APEX-SWAT is a central component of the U.S. Department of Agriculture (USDA) Conservation Effects Assessment Program. This program was initiated to support the objectives of the 2002 Farm Bill, and emphasizes the use of simulation modeling to quantify the environmental benefits of conservation practices at the national scale (Santhi et al. 2005). The APEX-SWAT integration was initially conducted in the context of the EPA-funded “Livestock and the Environment: A National Pilot Project (NPP)” initiated in 1992 to evaluate livestock contributions to environmental problems at the watershed scale (Gassman et al. 2005). The respective models were integrated within the “Comprehensive Economic Environmental Optimization Tool – Livestock and Poultry” (CEEOT-LP) modeling system developed within the context of the NPP (Gassman et al. 2002; Osei et al. 2000a; Osei et al. 2000b). The basic approach used by APEX-SWAT is to simulate the impacts of specific soil and nutrient management practices at field scale using APEX, and to input these edge-of-field sediment and nutrient loadings (dissolved and advected) into SWAT. SWAT subsequently routes these loadings through the stream network, within which their impact on ambient water quality can be evaluated (Gassman et al. 2005).

APEX

APEX is an evolution of the EPIC (Erosion-Productivity Impact Calculator; subsequently Erosion-Policy Impact Climate) model (Williams et al. 1984). EPIC was developed from 1981 onward and used extensively in the context of the Second Resources Conservation Act Appraisal (1985). EPIC contains nine distinct simulation components: weather, hydrology, erosion, nutrients, soil temperature, plant growth, plant environmental control, tillage and economic budgets. EPIC was designed as a field-scale model, where a “field” is conceptualized as a parcel that is relatively homogeneous with respect to weather, soil characteristics, topography, cropping patterns, and land management practices. It has been

used extensively in a wide range of studies examining soil erosion and conservation, crop yield response to inputs and management, nutrient and carbon cycling, water quality, and economic analysis of conservation policies and practices (Gassman et al. 2005). EPIC is a continuous simulation model operating at daily time steps. Soil erosion is calculated using one of nine established procedures, including the Universal Soil Loss Equation (USLE; Wischmeier and Smith 1978), Modified Universal Soil Loss Equation (MUSLE; Williams 1995) and Revised Universal Soil Loss Equation (RUSLE; Renard 1997) for water (rainfall, hydraulic) erosion; and the Wind Erosion Stochastic Simulator (WESS; Potter et al. 1998) for wind erosion.

APEX was developed primarily to enable the simulation of "farms" consisting of several "fields" as defined above, permitting the physical and economic evaluation of farming systems that are heterogeneous with respect to both physical and spatial attributes and management practices. APEX utilizes EPIC procedures at field level, and combines these to enable, for example, the evaluation of sediment and nutrient runoff as generated within one plot and attenuated by a second, down-slope plot containing filter or buffer strips. Like EPIC, APEX is a continuous simulation model operating at daily time steps. It possesses, in addition, the capacity to route runoff and associated contaminants between sub-units.

SWAT

SWAT is a physical-conceptual model at watershed scale. It combines simulation components that are rigorously physical with others that are conceptual or empirical. SWAT is a continuous simulation model operating at daily time steps, and has been used successfully over a range of timeframes. Typical applications involve annual or multi-year simulations, but SWAT has also been used to simulate the impacts of global climate change over longer periods. It is a comprehensive watershed simulation modeling system, integrating climate, hydrology, soil dynamics, biophysics, and water quality simulation capabilities and encompasses both land-phase watershed processes and receiving waters. SWAT, developed by the USDA Agricultural Research Service (ARS) (Neitsch et al. 2002, 2005), represents an evolution of the Simulator for Water Resources in Rural Basins (SWRRB) model (Arnold et al. 1995) and integrates components and algorithms from numerous pre-existing ARS models. These include Chemicals, Runoff and Erosion from Agricultural Management Systems (CREAMS; Knisel 1980), Groundwater Loading Effects on Agricultural Management Systems (GLEAMS; Leonard et al. 1987), and EPIC (Williams et al. 1984). Current versions include SWAT 2000 (2001), integrated within EPA's BASINS 3.0 framework, and SWAT 2005, the most recent version. BASINS is a multipurpose environmental analysis system that integrates a geographic information system (GIS), national watershed data, and a range of environmental assessment and modeling tools (USEPA 2008a).

The typical simulation domain of SWAT is a large watershed; from thousands of acres to thousands of square miles. Watersheds are partitioned into sub-basins, each of which is a mass balance (e.g., water, sediment, nutrient) accounting framework. Runoff, sediment, nutrients, and other associated constituents are generated within the sub-basin according to physical process models, and are routed to the sub-basin tributary via algorithms that reflect, among other factors, travel time and distance. From that point, they enter the model channel routing model subsystem, which simulates process dynamics for individual water quality constituents. There are no rigid or uniform criteria for subdividing watersheds into sub-basin accounting units, although this is typically done on the basis of similarity in land use, climate, and related factors. Similarly, there is no general restriction on sub-basin size distribution. A given SWAT model implementation may consist of large, coarsely parameterized units mixed with smaller, more detailed units.

Sub-basin surfaces are in turn subdivided into hydrologic response units (HRU), defined as unique soil/land use/management combinations within the sub-basin. The HRU is the spatial unit for which the land phase of the hydrologic cycle is simulated. Simulated outputs from each HRU (e.g., runoff, suspended sediment, nutrients) are summed over the sub-basin and introduced to the sub-basin channel, without regard to the spatial location of the HRU within that sub-basin. For situations in which it is important to take into account the overland flowpath of water and associated loadings, sub-basins can be further resolved into hill slope elements, which are connected via well-defined flowpaths as specified in a routing table. Hill slope elements are sub-basins (rather than HRUs), and as such are autonomous water and constituent accounting units. Hill slope discretization is used in evaluating the effectiveness of filter strips. It is also possible to introduce a grid system in areas of special interest.

SWAT consists of numerous simulation modules, described in detail in the SWAT Theoretical Documentation (Neitsch et al. 2002, 2005). Within the general sub-basin (HRU)–channel network configuration, SWAT simulation modules include climate (including a weather generator); surface runoff; evapotranspiration; soil water balance accounting; groundwater; nutrient and pesticide accounting (N, P dynamics, bacteria); erosion and sediment transport; crop growth; agricultural land and water management; urban runoff and contaminant wash-off; channel routing of water, sediments, nutrients, bacteria, and heavy metals; and impoundments, including regulated and unregulated reservoirs, ponds, and wetlands.

Surface runoff is simulated via two options. The first is the SCS Curve Number procedure (SCS 1972), which predicts runoff as a function of daily rainfall, hydrologic soil group, land management practice, and antecedent moisture conditions. The SCS curve number is a widely used empirical procedure, and is useful for simulating changes in runoff associated with changes in any of the conditioning factors in a computationally efficient manner. The second approach is based on the Green and Ampt (1911) infiltration model, which involves an idealization of the complex physical process of soil water infiltration. Precipitation is presumed to generate runoff at a rate equal to the difference between rainfall intensity and calculated infiltration capacity. Peak runoff rate, important in the simulation of soil erosion, is estimated via the rational method, where peak runoff is a function of rainfall intensity, watershed contributing area, and time of concentration. For larger sub-basins, SWAT utilizes a surface runoff lag procedure to simulate the overland routing of runoff to the sub-basin channel.

Erosion of soil is simulated in SWAT using the MUSLE (Williams 1995), an evolution of the widely used USLE (Wischmeier and Smith 1978). It is expressed as:

$$SED = 11.8\left(Q_{surf} \cdot q_{peak} \cdot A_{HRU}\right)^{0.56} \cdot K_{USLE} \cdot C_{USLE} \cdot P_{USLE} \cdot LS_{USLE} \cdot CFRG\,, \qquad \text{(Eq. 6-1)}$$

where *SED* is the daily sediment yield (MT), Q_{surf} is surface runoff volume (mm/Ha), q_{peak} is peak runoff rate (m^3/s), A_{HRU} is the HRU area (Ha), *K* is the USLE soil erodibility factor (empirical), *C* is the USLE cover and management factor, *P* is the USLE support practice factor, *LS* is the USLE topographic factor, and *CFRG* is the soil coarse fragment factor. Peak runoff and runoff volume are obtained as described above, and other components are calculated within SWAT on the basis of fixed (soil) and temporally varying (land cover, management) variables. In the land phase of SWAT, sediment moves with surface runoff, subject to the same lags for larger spatial units.

Within the channel network component of SWAT, water is routed to the basin outlet on the basis of either the variable storage routing or the Muskingum routing methods, respectively. In each case, rates and velocities are determined by Manning's formula. Channel flows are subject to transmission and evaporation losses, and to bank storage. SWAT versions through 2000 simulated sediment routing

through channels on the basis of stream power theory. In SWAT 2005, a simplified form of channel routing based on peak channel velocity is used. SWAT simulates four basic types of impoundments. Ponds, wetlands, and depressions (potholes) are by definition located within sub-basins and do not impound water flowing within the main channel system. Reservoirs are located on the main channel network, and each is subject to a separate water and sediment balance. For reservoirs, release can either be stipulated as a schedule, or releases can reflect daily, monthly, or average measured values. Outflow from ponds is calculated as a function of "target" storage, which varies seasonally. Wetlands and depressions release water whenever simulated volume exceeds capacity, or through release operations and engineered drainage (e.g., tiles). All impoundment structures can act as sediment traps. Sediments entering impoundments with inflow are assumed to be completely mixed, and subject to settling. Settling occurs when sediment concentrations exceed equilibrium values, expressed as megagram per cubic meter (Mg/m^3) of detained storage. Settling is simulated by the following equation:

$$C_{sed,f} = \left(C_{sed,i} - C_{sed,eq}\right) \cdot \exp^{\{-k_s \cdot t \cdot d_{50}\}} + C_{sed,eq}, \qquad \text{(Eq. 6-2)}$$

where $C_{sed,f}$ is the final sediment concentration (Mg/m^3), $C_{sed,i}$ is the initial concentration (Mg/m^3), $C_{sed,eq}$ is the equilibrium concentration (Mg/m^3), k is a decay constant (day^{-1}), t is the model time step (1 day), and d_{50} is the median particle size of inflow sediment (μm).

SWAT is intended to provide a comprehensive toolbox for predicting the impacts of land management decisions on water, sediment, nutrient, pesticide, and bacteria yields from predominantly rural and agricultural watersheds, and for Total Maximum Daily Load (TMDL) analysis as a component of BASINS. SWAT was designed to simulate large basins, but has been used effectively at a range of scales, from field to major watershed. SWAT models have been implemented for the Upper Mississippi Basin (Gassman et al. 2003) and over the entire coterminous United States, at basin scale. SWAT's quasi-physical specification is designed to enable simulation of ungauged basins, although in practice, the majority of published SWAT studies have involved calibration and validation on the basis of physically measured data. SWAT's design and functionality enable the linkage of upland land management activities influencing erosion with receiving water quality (sediment delivery) inclusive of channel routing and storage. With respect to BMPs, SWAT by design possesses the capability to simulate (1) erosion control via soil cover and management factors, (2) runoff control via flowpath modifications, and (3) sediment control via settling basins (ponds, reservoirs) and vegetated buffers (filter strips).

6.5.4 Dallas-Fort Worth Study Setting

Dallas-Fort Worth was selected as the location for an APEX-SWAT case study, primarily on the basis of an established SWAT model of the Trinity River watershed containing Dallas. Dallas, located at approximately 32°, 47' N Lat.; 96°, 48' W Lon.; and 131 m elevation in Northeastern Texas, is a major metropolitan area occupying roughly 385 mi^2 and containing a population of approximately 1.2 million (2006). Dallas is located within the Trinity River basin (USGS 08057000) in a generally low-relief setting. The Trinity is contained within 15 m levees through much of downtown Dallas, although efforts are currently underway to redevelop the urban river corridor as parkway and recreational facilities. Dallas has a humid sub-tropical climate, receiving around 37 inches (940 mm) of precipitation annually, with springtime (March–June) the rainiest period. Severe thunderstorms are common, particularly in spring and early autumn. Average temperatures range between 55 ^{0}F in January and 96°F in July. The Trinity River drains 6,106 mi^2 above Dallas. Annual average discharge ranges between 115 cubic feet per second (cfs) (1956) and 7,154 cfs (1982), averaging 1,844 cfs for the 1931–2007 period (USGS 2008). The East

Fork of the Trinity, containing one of the sample sub-watersheds, lies to the east of Metro Dallas and joins the Trinity River south and east of Dallas County.

The selection of sub-catchments within the Trinity and East Fork Trinity watersheds for high-resolution modeling in APEX-SWAT was guided primarily by three criteria. First, sub-watersheds must contain locations of recent significant urban development. Second, developed regions should not receive extensive inflow from upstream of imputed urbanizing areas, as this would tend to obscure local contributions to overall sediment (TSS) loadings. Finally, candidate sub-catchments should drain freely to the main stem of the Trinity (or primary tributaries); and not to lakes, reservoirs, or other impoundments that would complicate the simulation of sediment loadings downstream. Location and extent of urban development are of primary significance in this process. The Multi-Resolution Land Characteristics Consortium (MRLC) has developed comprehensive National Land Cover Databases (NLCD) for 1992 and 2001, respectively, derived from the Landsat Thematic Mapper-Extended Thematic Mapper imagery at 30-meter resolution. The 1992 NLCD coverages were re-projected and re-classified to achieve consistency with the 2001 coverages at 30-meter pixel level. Locations of land cover conversion from nonurban to urban uses, and from pervious to impervious surface characteristics, were identified via pixel-by-pixel comparison between the 1992 and 2001 datasets, respectively. Converted pixels serve to identify areas where construction activities occurred within the 1992–2001 period. This dataset, identified as the "NLCD 1992/2001 Retrofit Land Cover Change Product," is available at the MRLC Web site (http://www.mrlc.gov). As nonurban to urban land conversion consists of (1) agriculture-to-urban, (2) forest-to-urban, (3) grassland/shrub-to-urban, and (4) open water to urban categories, respectively, it should be noted that this procedure omits at least one category of construction activity, resulting when previously developed land is re-converted or re-developed. Thus, the procedure described identifies only nonurban to urban land conversion.

6.5.5 Sources of Data

General datasets required for successful SWAT implementation include climate (e.g., precipitation, temperature, radiation, humidity, wind speed); topography; soils; land use/land cover; stream reach network; and, ideally, data on stream discharge and water quality to enable calibration and validation. Most of these data are available from various sources including USGS hydrologic unit boundaries, topographic data at 30-meter resolution from the USGS National Elevation Dataset, soils data from the Soil Survey Geographic (SSURGO) database, and 1:24,000 land cover data from USDA's Natural Resources Conservation Service. Alternatively, land cover data from NLCD 2001 are available (MRLC 2001). Gridded monthly climate data (e.g., precipitation, maximum and minimum temperature, dewpoint) are available from the Parameter-Elevation Regression on Independent Slopes Model (PRISM; OSU 2007) for the entire United States. Monthly data are in general too coarse for most SWAT applications, and daily or finer resolution data for specific stations are available from the NOAA National Climatic Data Center. Weather data can also be simulated within SWAT, although an appropriate basis for calibration of the weather generator is required. Measured discharge data are available from USGS, and water quality data from USGS and EPA's STORET environmental data system, a repository for water quality, biological, and physical data (USEPA 2008f). Data for specific locations of construction activities will be gathered on a case-by-case basis, and probably represent the binding constraint to data acquisition for detailed SWAT simulations of construction activity.

6.5.6 Experimental Design

The basic methodological approach to evaluating regulatory options for construction site sediment abatement consists of (1) simulating a water quality baseline, and subsequently (2) generating alternative scenarios corresponding with specific sediment abatement options. The baseline assumes that construction activities occur at locations within the target sub-basins identified from the NLCD 1992/2001 Retrofit Land Cover Change Product ("retrofit product"). The retrofit product, overlaid on the local flow network and sub-watershed boundaries, appears in *Figure 6-6* (Trinity River within Dallas County) and *Figure 6-7* (East Fork Trinity River). Because areas of nonurban to urban land conversion span the 1992-2001 period, an assumption is made that the progress of development was uniform over this period. Assuming that approximately 10 percent of the developed land identified in the retrofit product is subject to development in a given year during this period (based on a further assumption that soils on developed parcels are exposed to erosive forces for one year or parts of one year only), an additional set of assumptions is required to estimate the distribution of areal extent of such developed parcels. This is accomplished on the basis of a construction site parcel size distribution discussed in EPA's *Development Document for Proposed Effluent Guidelines and Standards for the Construction and Development Category* (2008). Locations of target parcels within the sample sub-watersheds are selected to represent a range of settings, with emphasis on distance to watercourse and, to the extent applicable, slope, topography and soils. For the baseline simulation, no sediment remediation is assumed, so that the extent of sediment mobilization and delivery are determined by soil and other site characteristics, topography, rainfall erosivity, and related physical factors.

Once the sample parcels have been specified, the baseline simulation consists of a 30-year continuous simulation using meteorological records from the study area, augmented by the SWAT weather generator as required. Loadings of sediment, measured as Total Suspended Solids (TSS), are recorded at the outlets of each of the sample sub-catchments; and at representative points along the main stem of the Trinity River above and below the points of confluence. Under the baseline simulation, the locations and characteristics of sample parcels remain unchanged, since the objective is to evaluate the impacts of weather, season, and antecedent soil moisture conditions, respectively. Each successive scenario then involves the substitution of a reduced sediment loading, corresponding to a specific regulatory option (see description of options in *Chapter 5*.

A useful output is the spatial distribution of water quality (loadings) impacts, which will reflect both the spatial distribution of construction activities, the impacts of specific regulatory options, and other factors contributing to the background sediment loadings within the larger watershed.

Figure 6-6: Land Conversion Watershed Boundaries and River Network, Dallas County

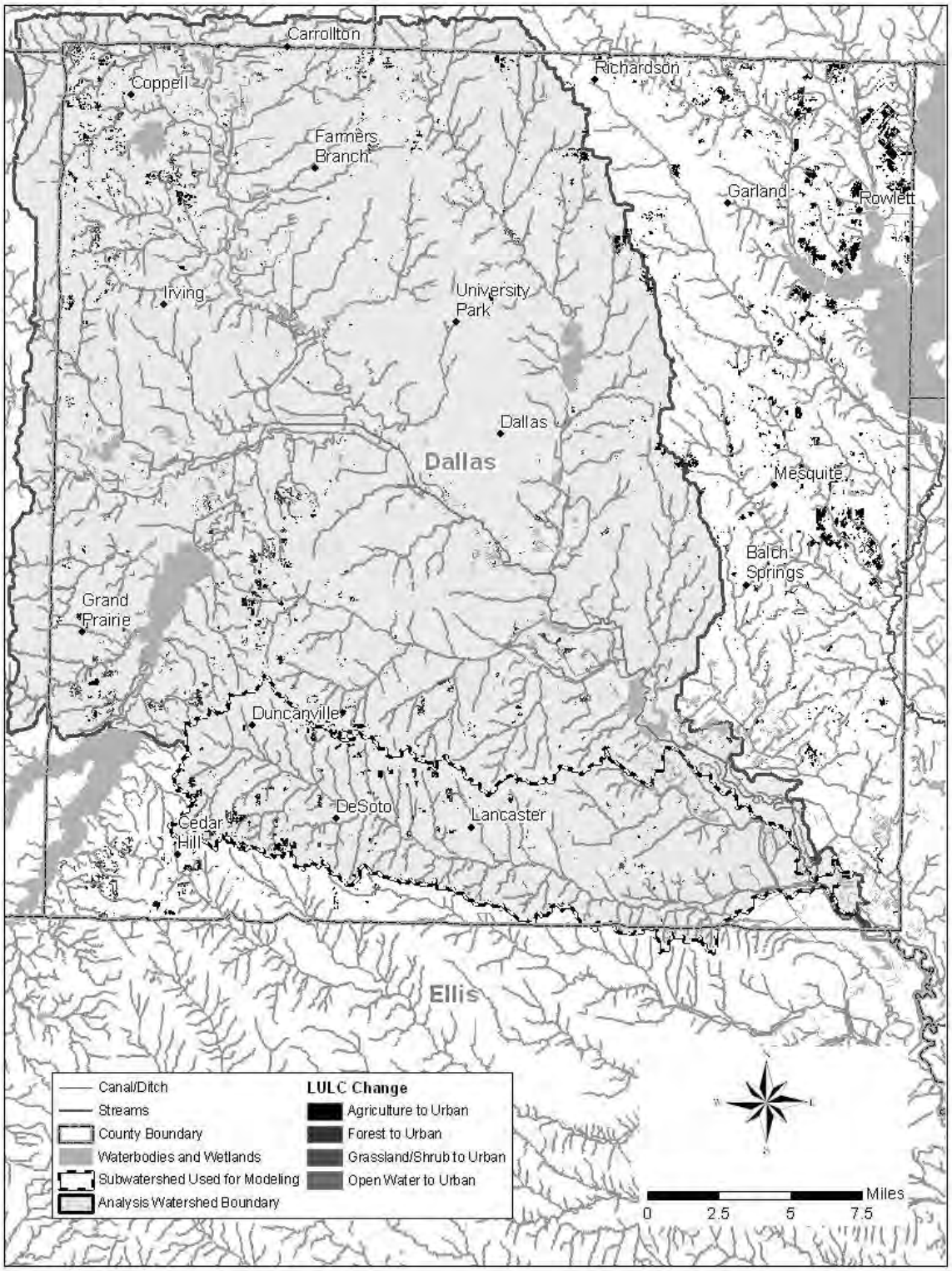
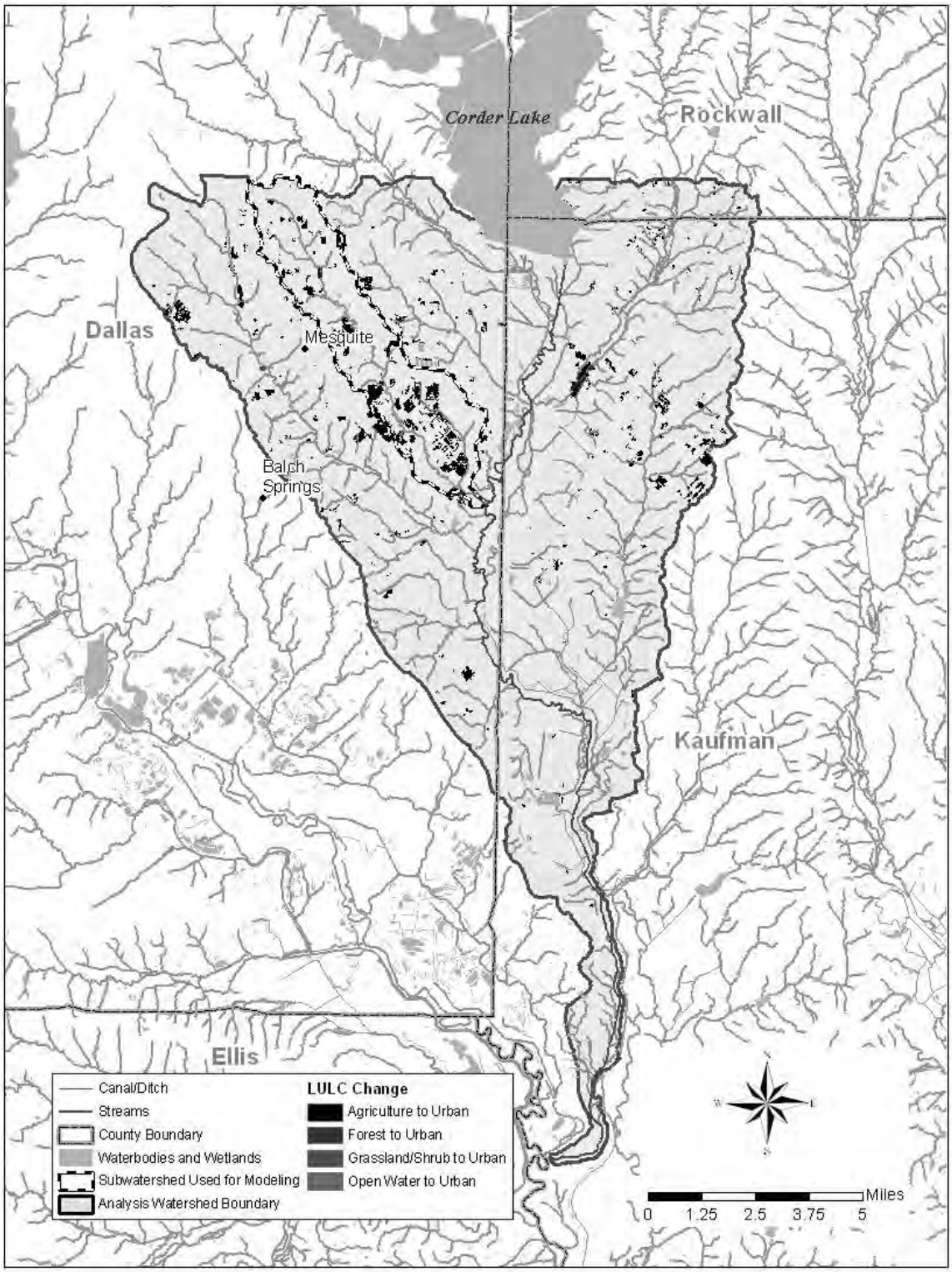


Figure 6-7: Land Conversion Watershed Boundaries and River Network, East Fork, Trinity River



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 2007).

Implementation of the C&D regulation is expected to result in less frequent dredging of navigable waterways due to reduced sediment runoff from construction sites and. This will result in cost savings 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).

In addition to the avoided costs when the need for dredging is reduced, there are also environmental benefits. Dredging is itself an environmentally disruptive activity 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, reintroducing the possibility of their uptake by fish and other aquatic organisms. Dredged sediment may be disposed of in another section of the water body or watershed, relocating the problem rather than removing it. Additionally, if the sediment removed from a site is contaminated, it could add to pollution at the disposal site (Clark et al. 1985). 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 C&D 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 proposed C&D regulation. The dredging database catalogs all USACE dredging contracts from 1990 to 2006 and all USACE-conducted dredging jobs from 1995 to 2006, 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 in the same EPA region. For each region, the Agency calculated 10^{th}, 50^{th}, and 90^{th} percentile costs and quantity boundaries and used these to fill in all incomplete records in that region. EPA included 50^{th} 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 10^{th} and 90^{th} 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 1995–2006, including dredging performed by USACE and dredging contracted to other firms. The cost data were adjusted to 2008$ using the construction costs of the producer price index (PPI) (Construction Cost Index 2008). During this period, USACE funded more than 3,700 dredging occurrences, worth more than $8.4 billion (2008$). This dredging removed more than 2.4 billion cubic yards of sediment from more than 1,100 navigable reaches. The region with the most dredging was Region 4, reporting more than 1,000 dredging occurrences between 1995 and 2006, nearly 100 per year. Region 6 reported the largest volume of sediment removed, with more than 1.3 billion cubic yards removed from navigable waterways over this 12-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, 1995–2006

EPA Region	Number of Dredging Occurrences	Total Sediment Removed (millions of cubic yards)	Total Cost (millions of 2008$)
1	75	3	$40
2	307	106	$754
3	283	113	$800
4	1,037	510	$2,954
5	703	104	$416
6	801	1,370	$2,466
7	21	11	$25
8	1	1	$2
9	134	66	$442
10	352	152	$551
Total	**3,714**	**2,435**	**$8,451**

Source: Dredging Information System (USACE 2007)

Table 7-2 summarizes dredging activity from 1995 through 2006 by type of entity performing dredging work (private or government). As noted above, USACE spent more than $8.4 billion (2008$) over the past 12 years to fund more than 3,700 dredging occurrences. The majority of dredging work (57 percent) was contracted out, with USACE performing about 43 percent of the work over this period. The individual costs of these dredging occurrences range considerably, from less than $10,000 to more than $100 million, but averaging about $2.6 million (2008$).

Table 7-2: Dredging of Navigable Waterways Performed or Contracted by USACE, 1995–2006

	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	1,613	43.4%	485	19.9%	$1,828	21.6%
Contract	2,101	56.6%	1,950	80.1%	$6,622	78.4%
Total	**3,714**	**100.0%**	**2,435**	**100.0%**	**$8,451**	**100.0%**

Source: Dredging Information System (USACE 2007)

7.2 Identifying Waterways That Are Dredged and the Frequency of Dredging

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 Frequency of Dredging

Continuous sediment deposition in a navigable waterway is likely to require repeated dredging to maintain navigability. Between 1995 and 2006, 1,166 sites required dredging on at least one occasion, and 475 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 (12 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 12 years by the number of occurrences of each job over these 12 years.[6] The EPA-estimated regional average recurrence intervals range from 8 to 12 years, including jobs that occurred only once in 12 years. For all dredging jobs in the United States, the average frequency of recurrence is approximately 9 years. Excluding jobs that occurred only once over this period, the regional average recurrence interval ranges from 3.5 to 6 years, averaging slightly less than 4 years at the national level.

Prior studies (Clark et al. 1985; Ribaudo 1989) of the cost savings 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 estimated based its estimation of reduced dredging activity based on historic dredging data from USACE. For dredging jobs that occurred only once between 1995 and 2006, 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)*.

[6] 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.

Table 7-3: Dredging Jobs and Recurrence Intervals, 1995- 2006

EPA Region	Number of Jobs	Number of Recurring Jobs	Average Interval (years)	Average Interval for Recurring Jobs (years)
1	41	10	10.0	3.8
2	92	43	8.2	3.9
3	118	42	9.3	4.5
4	268	100	8.8	3.5
5	277	126	8.4	4.0
6	211	96	8.1	3.5
7	16	1	11.6	6.0
8	1	0	12.0	N/A
9	48	20	8.6	3.8
10	94	37	8.8	4.0
Total	**1,166**	**475**	**8.7**	**3.8**

Source: USACE Dredging Information System (2007)

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 12 years. For the 691 jobs that only occurred once from 1995 to 2006, 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 12 years by the number of occurrences of that job between 1995 and 2006 to obtain an average interval of dredging for that job in years.

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 more than $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	$44	3.9	$11.11
2	$930.7	131.6	$7.07
3	$932.4	137.5	$6.78
4	$3,449.4	639.9	$5.39
5	$541.4	120.8	$4.48
6	$2,938.0	1,644.2	$1.79
7	$26.0	10.9	$2.39
8	$2.3	0.5	$4.49
9	$500.6	76.3	$6.56
10	$599.6	162.1	$3.70
Total	**$9,963.8**	**2,927.7**	**$3.40**

Source: USACE Dredging Information System (2007)

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.

7.4 Estimating the Total Cost of Navigable Waterway Maintenance Under the Baseline and Post-Compliance Scenarios

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. Using SPARROW output, EPA first determined sediment settlement in each state under the baseline scenario (no regulation in place), and then for the different policy options. The amount of sediment dredged in a state under a particular post-compliance scenario was divided by the amount dredged in the baseline scenario to obtain the percentage of sediment requiring dredging in that state under the post-compliance scenario. This calculation was performed at the state level rather than the reach level to adjust for any error in assigning river reach identifiers to dredging jobs.

EPA multiplied this statewide percentage by the average quantity of sediment dredged by each dredging job in a state to obtain an estimate of the amount of sediment requiring dredging by each occurrence of that job in the post-compliance scenario.

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 12 years from 2009 to 2021 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: Estimating the Reduction in Sediment Dredging in Navigable Waterways Due to the Reduction in Discharge from Construction Sites.*

Because costs will occur whenever the water body 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 12-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 12-year period for each job by dividing 12 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:

$$PV = \sum_{n=1}^{12/I} \left[\frac{Q_b * R_{pc} * C}{1 + (d * I)^n} \right] \qquad \text{(Eq. 7-1)}$$

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

$12/I$ = 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[7]

$d*I$ = 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.

[7] EPA estimated costs using both 3 and 7 percent discount rates, in accordance with OMB (2003, 33-34).

For example, if the Mississippi River is scheduled to be dredged every 2 years between 2009 and 2021, 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 2021, the Mississippi River will have been dredged six 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 12 years. Repeating this process for every RF1 reach in this analysis yields an estimate of national dredging costs over the next 12 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: Identifying Waterways That Are Dredged and the Frequency of Dredging* 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
  - Assigning 10th percentile cost and cubic yards dredged estimates for jobs lacking these data
  - Excluding jobs with only one occurrence over the 12 years from 1995 to 2006
  - Discounting dredging occurrences from the end of each occurrence interval.
- 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 12 years from 1995 to 2006 under the assumption that these jobs or similar jobs will occur once over the next 12 years
  - Discounting dredging occurrences from the end of each period as in the low estimate.
- 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 2006,
  - Discounting dredging occurrence at the beginning of each period.

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 12-year cost projection in order to facilitate the comparison of these costs to other costs calculated for the analysis of the C&D 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 2009 to 2021 are expected to range between $516.8 and $530.5 million per year (annualized using a 7 and a 3 percent discount rate, respectively). The high and low estimates produce an overall range between $342.4 million and $1.1 billion. 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	0.1	0.2	0.4	$0.4	$3.0	$13.0	$0.4	$2.8	$15.3
2	3.7	6.9	13.1	$28.0	$52.2	$107.1	$27.4	$50.5	$120.2
3	6.4	8.7	12.4	$45.0	$59.7	$96.5	$44.3	$58.1	$108.8
4	28.1	37.4	56.1	$115.2	$191.1	$442.1	$113.9	$186.1	$485.5
5	5.9	7.9	9.1	$22.6	$31.0	$47.7	$22.2	$30.1	$53.3
6	66.3	83.1	115.2	$94.3	$141.9	$239.7	$93.4	$138.9	$263.0
7	0.1	0.8	0.9	$0.1	$1.7	$2.5	$0.1	$1.6	$3.1
8	0.0	0.0	0.0	$0.0	$0.2	$0.2	$0.0	$0.2	$0.3
9	4.1	4.8	7.1	$26.3	$31.4	$47.5	$26.0	$30.7	$52.3
10	3.3	5.7	10.9	$10.5	$18.4	$38.6	$10.4	$17.9	$43.0
Total	**118.0**	**155.5**	**225.2**	**$342.4**	**$530.5**	**$1,034.8**	**$338.1**	**$516.8**	**$1,144.8**

Source: USEPA analysis

7.5 Estimating the Cost Savings from Decreased Dredging of Navigable Waterways

The cost savings 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 C&D regulation. *Table 7-6, Table 7-7,* and *Table 7-8* present annualized avoided cost estimates for navigable waterway dredging 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.

Annualized savings from reduced dredging activity range from $690,600 to $61.5 million, with Option 2 representing a savings of $12.6 to $12.9 million in the midpoint estimate. EPA estimates that Region 4 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 this region, and a large percentage reduction in sediment runoff expected as a result of the C&D regulation. Region 6, though it does account for a large portion of dredging activity, does not represent as large a portion of the cost savings due to its substantially lower cost per unit of dredging ($1.79, compared to $5.39 in Region 4) and larger number of construction sites exempted by rainfall and soil type waivers. Due to the lack of significant dredging activity in Region 8, no benefits are expected in this region.

Option 1, a construction general permit requiring sedimentation basins on all sites larger than 10 acres and the implementation of best management practices on smaller sites, is EPA's least stringent policy option. It is predicted to produce a range of cost savings between $690,000 and $1.8 million. Under this policy option, Region 4 will benefit from the most significant savings, accounting for almost all of the benefits under this option. This option does not predict any savings in Regions 7, 8, or 9.

Option 2 imposes a turbidity standard on construction sites larger than 30 acres in areas that have an "R-factor" (see *Chapter 5* for more details) of greater than 50 and with soils containing greater than 10 percent small particles, and requires a construction general permit for all other sites over one acre. The turbidity standard would require on-site treatment of stormwater, which would reduce TSS and turbidity in discharges from sites meeting the criteria for the turbidity standard. This option will prevent an estimated 3.3 million cubic yards of sediment from entering navigable water bodies and requiring dredging. The midpoint estimate for cost savings under this option is between $12.6 and $12.9 million per year, ranging from $8.9 million to $23.8 million between the low and high estimates. EPA expects more than 99 percent of these benefits to accrue to Regions 1 through 6, as many areas in Regions 7 through 10 are exempt through soil type or rainfall waivers provided by this option. Region 4 again represents a significant portion of savings, though Region 6 is also expected to benefit from a reduction of more than $1.6 million per year in dredging costs.

Table 7-6: Annualized Reductions in Dredging and 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]	0.0	0.0	0.0	$0.1	$0.4	$1.8	$0.1	$0.4	$2.0
2	0.2	0.3	0.5	$1.2	$2.2	$4.5	$1.1	$2.2	$4.8
3	0.2	0.3	0.4	$0.7	$1.3	$2.5	$0.7	$1.3	$2.7
4	169.3	204.0	302.3	$686.6	$962.8	$1,615.8	$676.7	$937.2	$1,741.7
5	0.2	0.2	0.3	$0.7	$0.9	$1.5	$0.7	$0.9	$1.6
6	1.1	1.3	1.8	$1.3	$2.0	$3.6	$1.3	$2.0	$3.8
7[1]	0.0	0.0	0.0	$0.0	$0.0	$0.0	$0.0	$0.0	$0.0
8	0.0	0.0	0.0	$0.0	$0.0	$0.0	$0.0	$0.0	$0.0
9[1]	0.0	0.0	0.0	$0.0	$0.0	$0.0	$0.0	$0.0	$0.0
10	0.0	0.0	0.0	$0.0	$0.0	$0.0	$0.0	$0.0	$0.0
Total	**171.0**	**206.1**	**305.4**	**$690.6**	**$969.8**	**$1,629.7**	**$680.7**	**$943.9**	**$1,756.6**

1 Reductions in dredged sediment and costs in these regions are not zero, but not sufficiently large to show at this level of significant digits

Source: USEPA analysis

Table 7-7: Annualized Reductions in Dredging and 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	0.2	0.8	1.2	$1.3	$9.3	$39.0	$1.3	$8.8	$44.5
2	29.6	53.6	98.9	$214.1	$408.1	$816.1	$210.2	$395.0	$878.6
3	79.8	112.6	162.1	$519.5	$720.5	$1,172.9	$512.0	$700.7	$1,276.4
4	1,744.6	2,125.3	3,136.1	$6,851.2	$9,759.3	$16,889.0	$6,758.1	$9,505.3	$18,161.5
5	30.0	39.9	47.2	$109.6	$145.2	$206.9	$107.7	$141.0	$222.9
6	744.7	951.3	1,314.2	$1,104.8	$1,669.5	$2,682.8	$1,093.2	$1,632.0	$2,846.0
7	0.7	4.0	4.8	$0.5	$7.9	$11.7	$0.5	$7.3	$13.9
8	0.0	0.0	0.0	$0.0	$0.0	$0.0	$0.0	$0.0	$0.0
9	2.7	3.2	4.8	$17.8	$21.2	$31.1	$17.5	$20.7	$33.0
10	35.4	52.9	84.5	$95.5	$161.9	$283.8	$94.7	$157.2	$304.5
Total	**2,667.8**	**3,343.6**	**4,853.9**	**$8,914.4**	**$12,903.0**	**$22,133.3**	**$8,795.3**	**$12,568.1**	**$23,781.3**

Source: USEPA analysis

Table 7-8: Annualized Reductions in Dredging and Costs Under Option 3

EPA Region	Reduction in Sediment Dredged (millions 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	1.9	3.0	$3.2	$22.9	$97.7	$3.2	$21.7	$111.1
2	45.5	82.5	153.1	$330.5	$628.1	$1,255.1	$324.5	$607.8	$1,352.0
3	118.7	167.9	241.6	$764.8	$1,066.6	$1,740.5	$753.8	$1,037.0	$1,894.3
4	3,586.9	4,401.9	6,741.2	$15,463.9	$21,927.8	$36,849.6	$15,248.8	$21,345.9	$39,776.9
5	49.7	65.5	77.4	$215.9	$280.1	$396.2	$212.5	$272.7	$425.4
6	1,100.1	1,411.8	1,949.1	$1,649.5	$2,494.1	$3,985.9	$1,631.9	$2,437.2	$4,232.8
7	1.0	5.9	7.2	$0.8	$11.7	$17.4	$0.8	$10.9	$20.7
8	0.0	0.0	0.0	$0.0	$0.0	$0.0	$0.0	$0.0	$0.1
9	22.7	26.7	39.6	$147.2	$175.3	$257.5	$145.1	$171.7	$273.1
10	137.9	203.8	319.6	$367.3	$620.1	$1,069.6	$364.2	$602.2	$1,147.5
Total	**5,063.0**	**6,368.0**	**9,531.7**	**$18,943.2**	**$27,226.8**	**$45,669.6**	**$18,684.7**	**$26,507.1**	**$49,233.9**

Source: USEPA analysis

Option 3, which imposes a turbidity standard on all sites larger than 10 acres (in addition to the requirements of Option 1) and requires a construction general permit for others, is EPA's most stringent policy option. This option would require on-site treatment of stormwater, substantially reducing its contribution to in-stream TSS and turbidity. Cost savings from this action range between $18.6 and $49.2 million, with a midpoint estimate of $26.5 to $27.2 million. Region 4 is again the largest beneficiary from this regulation, with $21.3 to $21.9 million in savings for the midpoint estimate, and Regions 3 and 6 also benefit from $1.0 to $1.1 million and $2.4 to $2.5 million in cost savings, respectively, under this option. The largest changes from Option 2 come in Region 9, which had many areas exempt due to rainfall under Option 2. Region 10 also sees a substantial increase in its benefits under this option, with the midpoint estimates rising from just over $157,200 and $161,900 (7 percent and 3 percent discount rates, respectively) under Option 2 to $602,200 and $620,100, respectively, under Option 3.

Overall, Regions 4 and 6 are the largest beneficiaries of reduced dredging costs from this action, both in terms of cost savings and in the amount of sediment prevented from entering and settling in navigable waterways. This is likely an effect of the large portion of coastline located in these regions and their large proportion of dredging activity.

7.6 Sources of Uncertainty and Limitations

- The USACE dredging database identifies dredging jobs by name, which is usually the name of the water body dredged. However, the data lack standardized naming conventions, so it is possible that the same water body 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). It is unclear whether this would lead to an over- or underestimate of benefits.
- The cost per cubic yard and interval data obtained from the dredging database vary significantly, even for different occurrences of dredging in the same water body or within regions. Aggregating such highly varied data and using regional averages may bias the cost and interval estimates. The direction of this bias is, however, uncertain.
- The analysis assumes a fixed cost per unit for dredging within each waterway, and that the rate of dredging (i.e., years between each dredging event) remains unchanged as a result of the C&D regulation. That is, the only change is assumed to be the quantity of sediment dredged during each dredging event. It is, however, possible that dredging frequency may be reduced in response to reduced sediment loads, rather than maintaining historical schedules with reduced quantities dredged during each event. If dredging frequency is reduced or if per-unit costs of dredging are nonconstant (e.g., there are fixed costs or nonconstant marginal costs of dredging), cost reductions may diverge from estimates presented above.

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 2007b).

The C&D 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 C&D 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: Review of Literature on Reservoir Sedimentation.*

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 increase its useful life
- 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 cost savings 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 conterminous 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: Water Quality Modeling*.

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. *Chapter 7, Section 7.3: Estimating the Navigational Maintenance Cost per Cubic Yard of Sediment Removed* 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: Water Quality Modeling* 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.[8] 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

[8] 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.

these frequencies as high and low estimates, and employed their arithmetic mean of 6 years for a midpoint 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: Estimating the Unit Cost of Sediment Removal from Reservoirs*. 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 442 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, and accordingly are estimated to have lower overall dredging costs. For other regions, midpoint estimates of dredging costs range between $50 and $255 million.

Table 8-2: Reservoir Dredging Under the Baseline Scenario

EPA Region	Sediment Dredged (yd^3)	Estimated Cost (millions of 2008$)		
		Low	Mid	High
1	1,294,949	$12.5	$13.3	$14.2
2	2,720,891	$16.8	$17.9	$19.0
3	5,361,722	$31.7	$33.7	$35.8
4	45,545,101	$214.2	$227.7	$241.9
5	35,464,668	$138.6	$147.4	$156.6
6	128,120,478	$199.7	$212.4	$225.6
7	41,412,208	$86.5	$92.0	$97.7
8	125,520,719	$491.4	$522.6	$555.0
9	41,840,133	$239.6	$254.8	$270.6
10	14,657,907	$47.3	$50.3	$53.4
Total	**441,938,775**	**$1,478.3**	**$1,572.0**	**$1,669.7**

Source: USEPA analysis

8.5 Estimating the Cost Savings from Reduced Reservoir Dredging

The difference between the anticipated dredging costs under the baseline and a particular policy option represents the cost savings of that policy option. *Table 8-3*, *Table 8-4*, and *Table 8-5* 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.

Cost savings from a reduction in reservoir sedimentation range from $470,000 to $32 million, depending on the policy options, the assumed frequency of reservoir dredging, and the discount rate. EPA's Option 2 represents a savings of $17.6 million for the midpoint estimate at a 3 percent discount rate and $15.9 million at a 7 percent rate. The largest savings are predicted in Region 4 under all options, as SPARROW predicts the largest overall reductions in sediment accumulation in this region. Region 4 accounts for nearly 98 percent of all savings under Option 1, and more than 50 percent in both Options 2 and 3. Region 6 is also anticipated to benefit from savings between $4.3 and $4.8 million under Option 2, and by between $6.5 and $7.4 million under Option 3 (at 3 and 7 percent discount rates, respectively).

Option 1, a construction general permit requiring sedimentation basins on all sites larger than 10 acres and the implementation of best management practices on smaller sites, is EPA's least stringent policy option. As noted above, Region 4 is the only region to benefit significantly from this policy option, and accounts for $593,000 of the $606,000 in cost savings for the midpoint estimate of this option at a 3 percent discount rate, and $536,000 out of $548,000 at a 7 percent rate.

Option 2 imposes a turbidity standard on construction sites larger than 30 acres in areas that have an "R-factor" (see *Chapter 3* and USEPA (2008b) for more details) of greater than 50 and have soils containing greater than 10 percent small particles, and requires a construction general permit for all other sites. The turbidity standard would require on-site treatment of stormwater, which would reduce TSS and turbidity in discharges from sites meeting the criteria for the turbidity standard. Cost savings under this option range between $13.7 and $18.7 million, with an expected value of $17.6 million at a 3 percent discount rate and $15.9 million at a 7 percent rate. This option greatly increases expected savings for all regions, reducing reservoir sedimentation rates by more than 100,000 cubic yards per year in five regions. Region 4 is predicted to realize a large portion of the cost savings, though Region 6 is also expected to benefit from $3.6 to $4.8 million dollars of reduced dredging costs each year. The next largest portion of savings

is predicted to accrue to Region 7, which will benefit by at least $1.2 million in the midpoint estimate. The Agency expects the smallest reductions in reservoir sedimentation in Regions 8 and 9, where many sites will be exempt due to rainfall or soil type waivers. Both of these regions will experience a decline of 10,000 cubic yards or less per year in the midpoint estimate of reservoir sedimentation.

Option 3, which imposes a turbidity standard on all sites larger than 10 acres (in addition to the requirements of Option 1) and requires a construction general permit for others, is EPA's most stringent policy option. This option would require on-site treatment of stormwater, substantially reducing its contribution to in-stream TSS and turbidity. Expected cost savings from this action range between $23.8 and $32.5 million, with midpoint estimates of $30.6 million at a 3 percent discount rate and $27.6 million at a 7 percent rate. Region 4 is again the largest beneficiary from this regulation, with approximately $16.4 million in savings for the midpoint estimate at a 3 percent discount rate and $14.8 million at a 7 percent discount rate. Region 6 is also estimated to benefit substantially from this regulation, with a reduction between $5.4 and $7.4 million. The drier Regions 8 and 9, which have many areas exempt under Option 2, are expected to benefit much more substantially under Option 3, with expected savings in Region 9 more than ten times greater under Option 3 than under Option 2 and nearly seven times greater in Region 8. Overall cost savings under this policy option nearly double for all estimate ranges.

Overall, Regions 4 and 6 are the largest beneficiaries of reduced dredging costs from this action, with substantial cost and sedimentation reductions also occurring in Regions 7 and 10 under Options 2 and 3. These two options are expected to result in significantly higher savings than Option 1, and are also anticipated to provide a more even distribution of these avoided cost benefits among regions.

Table 8-3: Reduction in Reservoir Dredging and 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	81	$0.8	$0.8	$0.9	$0.6	$0.8	$0.9
2	122	$0.8	$0.8	$0.9	$0.6	$0.7	$0.8
3	236	$1.4	$1.5	$1.6	$1.2	$1.3	$1.5
4	118,525	$557.3	$592.6	$629.4	$462.4	$535.9	$617.3
5	1,343	$5.3	$5.6	$5.9	$4.4	$5.0	$5.8
6	175	$0.3	$0.3	$0.3	$0.2	$0.3	$0.3
7	256	$0.5	$0.6	$0.6	$0.4	$0.5	$0.6
8	641	$2.5	$2.7	$2.8	$2.1	$2.4	$2.8
9	120	$0.7	$0.7	$0.8	$0.6	$0.7	$0.8
10	5	$0.0	$0.0	$0.0	$0.0	$0.0	$0.0
Total	**121,504**	**$569.5**	**$605.6**	**$643.2**	**$472.5**	**$547.6**	**$630.8**

Source: USEPA analysis

Table 8-4: Reduction in Reservoir Dredging and 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	9,405	$91.1	$96.9	$102.9	$75.6	$87.6	$101.0
2	11,408	$70.4	$74.8	$79.5	$58.4	$67.7	$78.0
3	77,481	$458.2	$487.3	$517.6	$380.2	$440.6	$507.6
4	1,894,111	$8,906.1	$9,470.5	$10,059.0	$7,389.6	$8,563.8	$9,864.6
5	151,379	$591.7	$629.2	$668.3	$491.0	$569.0	$655.4
6	2,754,706	$4,293.8	$4,565.9	$4,849.6	$3,562.7	$4,128.7	$4,755.9
7	701,561	$1,465.4	$1,558.3	$1,655.1	$1,215.9	$1,409.1	$1,623.1
8	8,869	$34.7	$36.9	$39.2	$28.8	$33.4	$38.5
9	3,541	$20.3	$21.6	$22.9	$16.8	$19.5	$22.5
10	182,545	$589.0	$626.3	$665.2	$488.7	$566.3	$652.3
Total	**5,795,007**	**$16,520.7**	**$17,567.7**	**$18,659.3**	**$13,707.7**	**$15,885.7**	**$18,298.7**

Source: USEPA analysis

Table 8-5: Reduction in Reservoir Dredging and 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	19,967	$193.5	$205.8	$218.5	$160.5	$186.1	$214.3
2	20,009	$123.4	$131.3	$139.4	$102.4	$118.7	$136.7
3	113,358	$670.4	$712.9	$757.2	$556.3	$644.7	$742.6
4	3,282,768	$15,435.5	$16,413.7	$17,433.6	$12,807.3	$14,842.2	$17,096.7
5	236,195	$923.3	$981.8	$1,042.8	$766.1	$887.8	$1,022.6
6	4,197,561	$6,542.8	$6,957.4	$7,389.7	$5,428.7	$6,291.3	$7,246.9
7	980,419	$2,047.9	$2,177.6	$2,313.0	$1,699.2	$1,969.1	$2,268.3
8	64,591	$252.9	$268.9	$285.6	$209.8	$243.2	$280.1
9	44,456	$254.6	$270.7	$287.5	$211.2	$244.8	$282.0
10	710,277	$2,291.6	$2,436.8	$2,588.3	$1,901.4	$2,203.5	$2,538.3
Total	**9,669,601**	**$28,735.9**	**$30,556.9**	**$32,455.6**	**$23,842.9**	**$27,631.3**	**$31,828.5**

Source: USEPA analysis

8.6 Sources of Uncertainty and Limitations

- 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.

Though sediment pools are built to accumulate sediment and preserve the useful capacity of the reservoir, they may also fill up more rapidly than anticipated at their initial construction, increasing the sediment build-up in a reservoir and increasing the cost of dredging it. It is also possible that these sediment pools themselves may be dredged. This analysis assumes that to maintain the current water storage capacity in the United States, all influent sediment will have to be removed in some manner, or replaced. Building replacement capacity is environmentally disruptive and may be more costly than sediment removal by dredging; therefore, this analysis assumes that dredging will be used to maintain reservoir capacity.

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 Sediment Effects on Drinking Water Treatment Costs

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
- 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.

9.2 Avoided Costs of Drinking Water Treatment from Reducing Sediment Discharge from Construction Sites

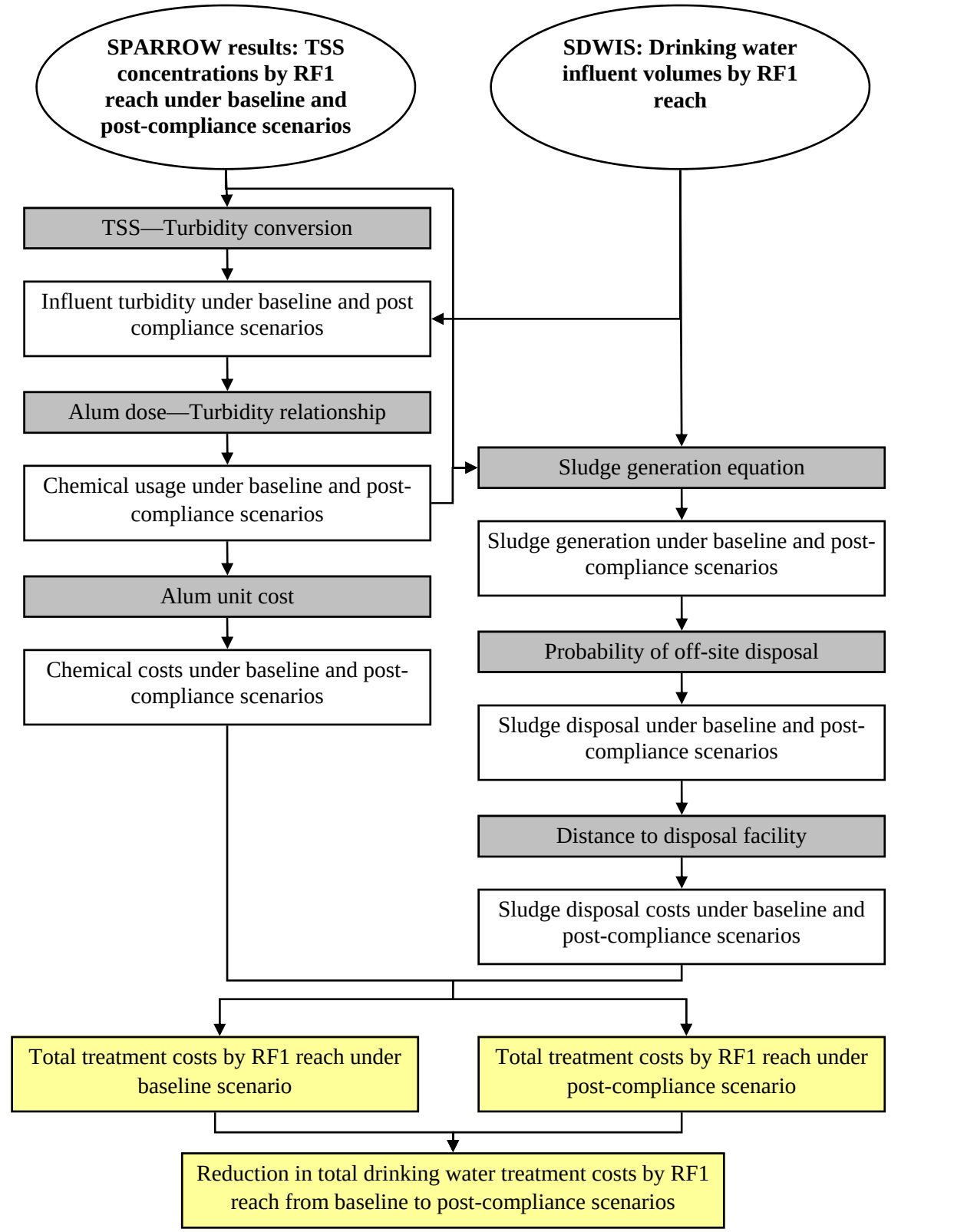
This section describes the estimation of the total expenditures to remove sediments from drinking water and the cost savings expected with the reduction of sediment discharge anticipated from the regulation. It involves the following steps:

- 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 cost savings between the baseline and post-compliance scenarios.

Figure 9-8 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 cost savings. 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: Sensitivity Analysis* for details).

Figure 9-8: 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 (EPA 2006a). 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 (Baruth 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 proposed rule (EPA 2000c). 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* (EPA 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 (EPA 2007b).

9.4 Modeling Sediment Concentrations and Reductions

EPA worked in conjunction with USGS to model sediment concentrations in RF1 stream reaches under both the current and proposed scenarios. A detailed description of this model, SPARROW, can be found in *Chapter 6: Water Quality Modeling*. SPARROW modeled the sediment concentrations in each RF1 reach according to many factors, including the amount of stormwater discharge from construction and development activities affecting that reach. SPARROW predicts sediment loads by suspended sediment concentration (SSC), which EPA converted into TSS by dividing the SPARROW-generated values by 1.3. This factor compensates for a 30 percent negative bias in TSS concentrations (Gray et al. 2000).

EPA assumed, as per Ngo and Nichols (1998), that facilities with baseline TSS concentrations exceeding 1,000 mg/L would employ 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 De Zuane (1997) and

Yao (1975) 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.[9]

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: Sediment Effects on Drinking Water Treatment Costs*.

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):

$$F = 0.067 * P^{1.062} \quad \text{(Eq. 9-1)}$$

Where:

F = average daily flow in thousands of gallons per day

P = population served

Table 9-1 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.

[9] 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 9-1: 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 (2007)

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 Baruth (2005) where TSS (mg/L) = b • turbidity (in NTU) where b has a value between 0.8 and 2.2. The value values assigned to b in this analysis are 2.2 for the low estimate, 1.5 for the midpoint estimate, and 0.8 for the high estimate (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 $373 per ton (2008$) (EPA 2005b). EPA's assumption that all turbidity will be treated with alum may understate treatment costs and savings. EPA estimated cost savings 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: Estimating the Amount of Sludge Generated,* 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:

$$Al = 33\log(T) - 28 \quad \text{(Eq. 9-2)}$$

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 $907x10^6$ 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.

$$TC_{Al} = Q * \left(\frac{Al * 3.8}{907 * 10^6} \right) * 365 * C_{Al} \quad \text{(Eq. 9-3)}$$

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 water body 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 water body)
- 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 (2007a) derived the following relationship between sludge generation and chemical coagulants:

$$S = 8.34 * Q * [2.0CaCH + 2.6MgCH + CaNCH + 1.6MgNCH + CO_2 + 0.44Al + 2.9Fe + SS + A] \quad \text{(Eq. 9-4)}$$

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 (1996a):

$$Y = \left[54.12\left(S * 365 * .0005 * \mathrm{Pr}\right)\right] + \left[(3.83 + 0.25Z)\left(S * 365 * .0005 * \mathrm{Pr}\right)\right] \qquad \text{(Eq. 9-5)}$$

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 + 0.25 = transportation cost ($)

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% 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% for all options
  - TSS = 0.8 * Turbidity
  - C_{Al} = $327

9.9 Estimating the Total Costs of Drinking Water Treatment Under the Baseline and Post-Compliance 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 9-2* presents annual estimated total drinking water treatment costs for the baseline scenario, including low, midpoint, and high estimates for avoided costs. National drinking water treatment costs for TSS and turbidity range from around $377 to about $742 million between the low and high estimates for the baseline scenario, with an expected value of

$585 million. Regional turbidity treatment costs range from $8.4 million in Region 10 to between $85 and $110 million in Regions 4, 5, 6, and 9 for the midpoint estimate.

Table 9-2: 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	43.1	63.9	121.1	$10.9	$16.5	$22.2
2	53.3	90.5	192.8	$28.3	$44.6	$58.5
3	73.3	107.9	203.3	$48.7	$65.8	$77.6
4	109.3	161.2	303.7	$61.4	$84.5	$100.8
5	139.2	274.8	647.7	$54.3	$88.2	$113.1
6	190.6	610.7	1,765.9	$64.8	$107.1	$139.9
7	167.1	424.8	1,133.6	$18.3	$33.7	$44.3
8	123.6	267.1	661.7	$18.3	$25.6	$30.6
9	136.1	323.4	838.4	$65.9	$110.5	$143.9
10	45.7	83.9	188.9	$5.5	$8.4	$11.1
Total	**111.0**	**247.9**	**624.5**	**$376.5**	**$584.9**	**$742.0**
Cost per million gallons[1]				**$59.4**	**$92.2**	**$117.0**

[1] Calculated for the estimated 17,373 million gallons per day of intake for public water systems in SDWIS.

Source: USEPA analysis

It is important to note that these avoided cost estimates do not take into account any treatment or disposal cost savings 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 Cost Savings from Lower Sediment and Turbidity in Drinking Water Influent

The total cost savings from lowered turbidity resulting from lower TSS concentrations in drinking water influent was estimated as the difference between drinking water turbidity treatment under the baseline and post-compliance scenarios. *Table 9-3* ,*Table 9-4* and *Table 9-5* present estimated savings in drinking water treatment costs from lowered turbidity for the three policy options, as well as low, midpoint, and high estimates for savings under each of these options.

The anticipated savings from reduced TSS and turbidity treatment at drinking water facilities are between $181,900 and $14.5 million, varying substantially between the least and most stringent policy options though less dramatically between the low and high estimates. Option 2 is expected to reduce TSS and turbidity treatment costs for drinking water facilities by between $5.8 and $8.0 million. As is the case with navigable waterway and reservoir dredging, Region 4 benefits most significantly and consistently from the TSS reductions expected from this regulatory action. The avoided costs in Region 4 under Policy Option 1 account for a significant portion of the total national savings. Other regions receive a larger portion of benefits under Policy Options 2 and 3, though Region 4 still accounts for the greatest proportion of the cost reductions. Regions 6 and 10 also show significant savings under Options 2 and 3, with savings in Region 10 between $1.2 and $3.5 million in all estimates for these two options.

Option 1, a construction general permit requiring sedimentation basins on all sites larger than 10 acres and the implementation of best management practices on smaller sites, is EPA's least stringent policy option. Region 4 receives the only considerable cost savings under this option, between $170,500 and $200,500. Savings in Regions 2, 5, and 8 are between $1,00 and $10,000, and in all other regions, expected cost reductions are less than $1,000.

Option 2 imposes a turbidity standard on construction sites larger than 30 acres in areas that have an "R-factor" (see *Chapter 5* for more details) of greater than 50 and with soils containing greater than 10 percent small particles, and requires a construction general permit for all other sites. The turbidity standard would require on-site treatment of stormwater, which would reduce TSS and turbidity in discharges from sites meeting the criteria for the turbidity standard. For the midpoint estimate of Option 2, the national average reduction in treated turbidity is 1.7 NTU, though in Region 4 (where many of the benefits are expected) the reduction is more than 5 NTU. Region 10 is expected to benefit from an average reduction of 4.3 NTU in influent water turbidity. The expected value of avoided costs for this estimate is $7.4 million. Estimated savings exceed $1 million in Regions 4, 6, and 10. These savings are much greater than the near-zero savings expected under Option 1 for Regions 6 and 10. Regions 8 and 9 do not benefit as substantially as other regions, likely due to a large number of sites being exempted from the turbidity standard.

Option 3, which imposes a turbidity standard on all sites larger than 10 acres (in addition to the requirements of Option 1) and requires a construction general permit for others, is EPA's most stringent policy option. This option would require on-site treatment of storm water, substantially reducing its contribution to in-stream TSS and turbidity. Total avoided costs for this option are between $10.4 and $14.5 million, with a midpoint estimate around $13.1 million. The drier Regions 8 and 9, which have many areas exempt under Option 2, are expected to benefit much more substantially under Option 3. Estimated avoided costs are seven to ten times larger under this option compared to Options 1 and 2, though still much lower than in other regions.

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 and 3 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 7.2 NTU nationwide, with a reduction of nearly 13 NTU in Region 4 and of nearly 12 NTU in Region 7. Under Option 3, the high estimate is on average around 11.2 NTU nationwide, 22.2 NTU in Region 4, 13.1 NTU in Region 6, and nearly 17 NTU in Region 7.

Overall, Regions 4, 6, and 10 receive the largest proportion of the cost reductions from reduced turbidity in drinking water. Options 2 and 3 produce significantly larger cost savings than Option 1, which is expected to reduce treatment costs by only about $211,400 nationally (midpoint estimate). Midpoint estimates of avoided costs under Options 2 and 3 are approximately $7.4 and $13.1 million, respectively. Under Options 2 and 3, cost savings are distributed more evenly among the 10 regions, though the aforementioned regions benefit most due to a more substantial effect of the regulation in these regions.

Table 9-3: Reduction in Drinking Water Treatment Costs Under Option 1

EPA Region	Average Reduction in Treated Turbidity (NTU) [1]			Cost Savings (thousands of 2008$)		
	Low	Mid	High	Low	Mid	High
1	0.0	0.0	0.0	$0.2	$0.2	$0.3
2	0.0	0.0	0.0	$4.2	$5.0	$5.0
3	0.0	0.0	0.0	$0.2	$0.2	$0.2
4	0.5	0.8	1.4	$170.5	$196.7	$200.5
5	0.0	0.0	0.1	$5.6	$7.8	$9.3
6	0.0	0.0	0.0	$0.1	$0.1	$0.1
7	0.0	0.0	0.0	$0.1	$0.1	$0.2
8	0.0	0.0	0.0	$1.1	$1.4	$1.5
9	0.0	0.0	0.0	$0.0	$0.0	$0.0
10	0.0	0.0	0.0	$0.0	$0.0	$0.0
Total	**0.1**	**0.1**	**0.2**	**$181.9**	**$211.4**	**$217.1**

[1] Average turbidity reductions shown as 0.0 are not actually zero, but not sufficiently large to show at this level of significant digits
Source: USEPA analysis

Table 9-4: Reduction in Drinking Water Treatment Costs Under Option 2

EPA Region	Average Reduction in Treated Turbidity (NTU) [1]			Cost Savings (thousands of 2008$)		
	Low	Mid	High	Low	Mid	High
1	0.1	0.1	0.2	$11.5	$14.4	$15.9
2	0.3	0.5	1.0	$137.1	$179.4	$197.9
3	1.4	2.0	3.8	$567.0	$652.7	$653.3
4	4.6	6.7	12.7	$2,070.0	$2,474.5	$2,583.8
5	0.6	1.0	2.3	$192.4	$258.5	$301.6
6	2.1	3.7	8.0	$1,369.0	$1,662.9	$1,738.3
7	2.9	5.3	11.8	$322.0	$411.8	$468.7
8	0.0	0.1	0.1	$4.9	$6.1	$6.7
9	0.1	0.1	0.2	$7.3	$10.1	$12.0
10[2]	4.9	15.9	46.2	$1,165.7	$1,750.6	$2,034.4
Total	**1.7**	**3.1**	**7.2**	**$5,846.9**	**$7,421.0**	**$8,012.6**

[1] Average turbidity reductions shown as 0.0 are not actually zero, but not sufficiently large to show at this level of significant digits
[2] The estimated turbidity reductions for Region 10 are heavily influenced by six drinking water treatment facilities on a reach with the estimated TSS concentration greater than 5,000 mg/L in the baseline. The estimated TSS concentration is reduced significantly post-compliance.
Source: USEPA analysis

Table 9-5: Reduction in Drinking Water Treatment Costs Under Option 3

EPA Region	Average Reduction in Treated Turbidity (NTU)			Cost Savings (thousands of 2008$)		
	Low	Mid	High	Low	Mid	High
1	0.4	0.6	1.1	$76.7	$93.1	$101.8
2	0.6	0.9	2.0	$279.0	$361.1	$394.5
3	2.0	2.9	5.5	$837.6	$964.8	$965.7
4	8.0	11.8	22.2	$4,398.0	$5,272.3	$5,422.2
5	1.0	1.8	4.0	$302.5	$407.6	$475.5
6	3.1	5.8	13.1	$2,161.7	$2,644.6	$2,772.5
7	4.1	7.5	16.7	$450.9	$581.7	$665.4
8	0.2	0.4	0.8	$29.9	$36.5	$39.1
9	0.6	1.0	2.1	$81.0	$112.2	$134.4
10[1]	7.7	21.8	60.6	$1,751.7	$2,667.5	$3,488.6
Total	**2.8**	**5.0**	**11.2**	**$10,369.1**	**$13,141.5**	**$14,459.9**

[1] The estimated turbidity reductions for Region 10 are heavily influenced by six drinking water treatment facilities on a reach with the estimated TSS concentration greater than 5,000 mg/L in the baseline. The estimated TSS concentration is reduced significantly post-compliance.

Source: USEPA analysis

9.11 Sources of Uncertainty/Limitations

- 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 cost savings of smaller quantities of sludge to be disposed of) for facilities that generate toxic sludge.
- 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 Benefits from Water Quality Improvements

As discussed in the preceding chapters of this document, sediments 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 original meta-analyses are presented in Appendix A as well as in sources such as Johnston et al. (2005; 2006), Bateman and Jones (2003), Shrestha et al. (2007), Rosenberger and Phipps (2007), and USEPA (2004c).

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 that can be more easily interpreted by policy makers and the general public. The parameters used in formulating the WQI are determined based on water body type, the 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.

The WQI used in this analysis is based on a methodology originally developed by McClelland (1974), which EPA subsequently modified to rely on fewer parameters (USEPA 2002, USEPA 2004c). For this analysis, EPA further modified the WQI methodology to apply to estuarine reaches, building on prior applications of WQI concepts to saltwater environments (Harrison et al. 2000; CCME 2001; Carruthers and Wazniak 2003; Gupta et al. 2003; and USEPA 2007c).

The WQI methodology used for rivers and streams and for estuaries are described in *Section 10.1.1* and *Section 10.1.2*, respectively.

10.1.1 WQI Methodology for Rivers and Streams

McClelland (1974) relied on a survey of water quality experts to identify water quality parameters, 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). Following McClelland's approach, EPA estimated the WQI for a given reach using a weighted geometric mean function as follows:

$$WQI_r = \prod_{i=1}^{n} Q_i^{Wi} \quad \text{(Eq. 10-1)}$$

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

Table 10-1 provides the parameters and their weights for inland rivers and streams (freshwater).

Table 10-1: Original and Revised Weights for Freshwater WQI Parameters

Parameters[1]	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$_5$ (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 (TSS(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**

[1] See Table 10-2 for detail on developing parameter quality values.

Source: McClelland 1974; USEPA 2002

To calculate the WQI, EPA first scaled the data parameters to a standard (0–100) range. The parameter quality values are based on parameter concentrations in the affected water bodies and quality curves developed by McClelland (1974) and updated by the Oregon Department of Environmental Quality for the Oregon Water Quality Index (OWQI) (Cude 2001; Dunnette 1979). *Table 10-2* presents parameter-specific functions used in developing parameter quality values (hereafter referred to as water quality subindexes).

Table 10-2: Freshwater Water Quality Subindex

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
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)			
TSS	< 40	mg/L	100
TSS	40 < TSS < 300	mg/L	124.7 * exp(TSS * -5.552 E-3)
TSS	> 300	mg/L	10
Biochemical Oxygen Demand, 5-day (BOD)			
BOD	≤8	mg/L	100 * exp(BOD * -0.1993)
BOD	>8	mg/L	10

Source: McClelland 1974; Cude 2001

10.1.2 WQI Methodology for Estuaries

The methodology used in calculating a WQI score for estuaries and coastal water bodies 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 2007c) to identify water quality parameters pertinent to estuaries and their relative importance in characterizing overall water quality.

The relevant parameters include parameters that are common to both freshwater and saltwater systems (dissolved oxygen, nitrogen, phosphorus, fecal coliform, and suspended solids). They also include chlorophyll-a (ChA). Chlorophyll-a assesses the amount of phytoplankton present in the water body. While phytoplankton is a component of the water body'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.

The weighting scheme used to combine the parameters is a modification of the one used for the freshwater WQI. Table 10-3 provides the parameters and their weights.

Table 10-3: 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 (SSC(mg/L))	0.11
Chlorophyll-a (mg/L)	0.08
Total	**1.00**

[1] Weights used for rivers and streams were redistributed to account for the removal of BOD and the addition of ChA.

Similarly to the WQI for rivers and streams, the data for each parameter are mapped to a standard (0–100) range using subindex curves. Table 10-4 presents parameter-specific functions used in developing water quality subindexes for estuaries. The transformation curves are based on the ones used for freshwater systems, with the exception of TSS, for which the curve is adjusted to account for the range of concentrations characteristic of estuarine conditions. The curve used for ChA is derived from the curves used for TN, since both parameters indicate trophic status.

Table 10-4: Estuarine Water Quality Subindex

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
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

Table 10-4: Estuarine Water Quality Subindex			
Parameter	**Concentrations**	**Concentration Unit**	**Subindex**
Total Suspended Solids (TSS)			
TSS	< 40	mg/L	100
TSS	40 < TSS < 550	mg/L	119 * exp(TSS * -0.0045)
TSS	> 550	mg/L	10
Chlorophyll-a (ChA)			
ChA	≤ 40	ug/L	100 *exp(ChA * -0.05605)0
ChA	> 40	ug/L	10

Source: McClelland 1974; Cude 2001

A similar correspondence between WQI and suitability for potential uses is then applied to relate the estuarine WQI score to use attainability.

10.1.3 Relation between WQI and Suitability for Human Uses

Vaughan (1986) used a reduced set of parameters in a reweighted equation to develop 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 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 reduced form of the WQI:

- The USGS National Water Information System (NWIS) provided concentration data for three parameters: (1) fecal coliform, (2) dissolved oxygen, and (3) biochemical oxygen demand.
- The USGS model outputs provided data for nitrogen and phosphorus concentrations, as well as suspended sediment concentrations. The suspended sediment concentrations were estimated for the baseline scenario and each policy option. EPA converted suspended sediments measurements

to TSS by dividing the SPARROW-generated values by 1.3. This factor compensates for a 30 percent negative bias in TSS concentrations (Gray et al. 2000).

- EPA's Storage and Retrieval (STORET) data warehouse provided additional data on fecal coliform counts and biochemical oxygen demand (EPA 2008e). The NWIS database was not available 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.

The following data sources provided data for estuaries and coastal water bodies:

- 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.
- 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.
- EPA's STORET data warehouse provided additional data on all relevant parameters, including fecal coliform counts, but excluding chlorophyll-a (EPA 2008e).

The analysis of baseline WQI is based on 57,531 reaches in the conterminous United States.[10] EPA removed from the analysis all reaches with TSS concentration in the 95th percentile or above, considering them to be outliers. Ambient concentrations of WQI parameters were available for a total of 9,444 reaches (9,122 of which were included in the analysis after removing all outliers). EPA used a successive average approach to address the data gaps in the remaining freshwater reaches. If ambient concentrations for a WQI parameter were available for a reach in a given HUC, the data were extrapolated to all reaches in that HUC under the assumption that reaches in the same watershed share similar characteristics. A successive average was calculated at the HUC8, HUC6, and HUC2 level for all reaches that did not fall into a smaller HUC level.[11] Using this estimation approach, EPA was able to compile water quality data for a total of 57,206 freshwater reaches (including 48,084 reaches estimated from HUC-based averages).

In the case of estuaries and coastal reaches, EPA used a similar approach to estimate missing average concentrations for reaches based on the average concentration at the HUC level. Thus, for RF1 reaches where no empirical measurement was found, the parameter value was estimated by averaging the corresponding values for adjacent estuarine/coastal reaches within the same HUC8 watershed. Due to the scale of estuarine water bodies, EPA used HUC8 to estimate average parameter concentrations and did not estimate values based on HUC 6 or HUC2. Using this approach, EPA compiled ambient water quality data for 2,068 coastal reaches in the conterminous United States; 325 of these reaches had data empirical data available at the HUC8 level and were included in the analysis.

[10] No ambient concentration data were available for Alaska and Hawaii.

[11] HUCs are catalogue numbers with 8 to 14 digits, with each set of 2 digits giving more specific information about the water. 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. 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 waters for which the first six digits of their HUC are the same.

EPA then calculated baseline WQI for all reaches based on measured or estimated parameters, and baseline TSS concentrations estimated in SPARROW. 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.[12] *Table 10-6* shows the distribution of river miles by WQI values and EPA Region for the baseline scenario (current conditions).

Table 10-6: Percentage of River Miles in Conterminous 48 States by WQI Classification for EPA Regions: Baseline Scenario

EPA Region	WQI<25	25≤WQI<50	50<WQI<70	70<WQI
1	0.18%	5.40%	72.82%	21.61%
2	0.33%	15.07%	30.36%	54.24%
3	4.70%	46.05%	43.50%	5.75%
4	1.21%	42.49%	39.71%	16.60%
5	8.95%	62.78%	24.60%	3.67%
6	2.55%	72.73%	19.95%	4.78%
7	28.03%	66.08%	5.49%	0.40%
8	5.05%	54.85%	25.48%	14.62%
9	19.69%	73.80%	5.97%	0.55%
10	0.08%	20.69%	30.30%	48.93%
National Average	**7.05%**	**52.25%**	**26.23%**	**14.46%**

Source: U.S. EPA analysis. 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 estimated the WQI for each policy option. In calculating the post-compliance WQI, the Agency used option-specific TSS concentrations from the SPARROW output. The sediment loading estimates for each policy option reflect the expected reduction in sediment discharge under the policy options. Although reductions in sediment loadings from construction sites may also reduce loadings of other pollutants that contribute to the WQI such as nutrients, the other contributing parameters to the WQI were held constant in this analysis for all policy options.

Each RF1 reach that has an improved WQI value from the baseline scenario to a policy option contributes to the estimated economic benefits. 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 even 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 river 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 ΔWQI≤0.1, 0.1<ΔWQI≤0.5, and

[12] "Terminal" reaches that terminate into the Great Lakes or along the Canadian border were treated as inland rivers and streams.

0.5< ΔWQI. For each combination of the baseline water quality category and the improvement range, the Agency estimated average ΔWQI and the corresponding percentage of total river miles in the state.

Table 10-7, *Table 10-8*, and *Table 10-9* summarize changes in ambient water quality resulting from Options 1, 2, and 3, respectively. *Appendix D* 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 9,325 RF1 miles or 1.6 percent of the 585,368 RF1 miles receiving construction discharges. EPA Region 4 is essentially the only geographic region that shows any significant water quality improvements, with 9,177 RF1 miles (10.5 percent of RF1 miles receiving construction discharges in this region) being affected. Region 5 is expected to have the next largest water quality improvement with 96 RF1 miles being affected under the post-compliance scenario.

EPA's Option 2 is estimated to improve ambient water quality in 11,418 (19.0 percent) RF1 miles that receive construction discharges nationwide. EPA's Regions 2, 3, and 4 are each estimated to experience improved water quality on 35 percent or greater of RF1 miles included in the analysis. EPA's water quality analysis predicts that Regions 3 and 4 will experience the most improvements in terms of both RF1 miles (18,844 and 48,229) and percentage of total regional reach length (64.6 percent and 55.2 percent). EPA's analysis also indicates that Region 4 would benefit the most in terms of the estimated magnitude of water quality improvements with 5.5 percent of RF1 miles that receive construction discharges estimated to improve by greater than 0.5 WQI units. Conversely, EPA Region 8 is estimated to improve the least, with 288 river miles benefiting from higher water quality.

Option 3 yields the most significant results overall with 136,488 RF1 miles expected to improve under the post-compliance scenario. The estimated scale of improvements ranges from 3.6 percent to 71.4 percent of RF1 miles in EPA's Region 8 and 3, respectively. EPA estimated that Region 4 will experience the largest water quality improvements, with 54,727 RF1 miles (62.7 percent of the RF1 miles that receive Construction discharges) affected. EPA's analysis also shows that Region 4 will have the most RF1 miles (8,486) improving by greater than .5 WQI units, while Region 9 will have the lowest water quality improvements with 2,193 miles showing any improvement under the post-compliance scenario.

Table 10-7: Estimated Water Quality Improvements Under Option 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		
	RFI Miles Receiving Construction Discharges	Miles of River in RF1 Network	River Miles	% of River Miles Receiving Construction Discharges	% of Total River Miles	River Miles	% of River Miles Receiving Construction Discharges	% of Total River Miles	River Miles	% of River Miles Receiving Construction Discharges	% of Total River Miles	River Miles	% of River Miles Receiving Construction Discharges	% of Total River Miles
1	16,952	18,293	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
2	15,888	16,110	20	0.1%	0.1%	0	0.0%	0.0%	0	0.0%	0.0%	20	0.1%	0.1%
3	29,183	33,617	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
4	87,312	94,464	8,207	9.4%	8.7%	738	0.8%	0.8%	232	0.3%	0.2%	9,177	10.5%	9.7%
5	71,508	71,550	96	0.1%	0.1%	0	0.0%	0.0%	0	0.0%	0.0%	96	0.1%	0.1%
6	82,855	98,669	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
7	54,018	60,909	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
8	113,710	130,311	32	<0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	32	<0.0%	<0.0%
9	47,474	56,259	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
10	66,468	69,495	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
National Total	**585,368**	**649,676**	**8,355**	**1.4%**	**1.3%**	**738**	**0.1%**	**0.1%**	**232**	**<0.0%**	**0.0%**	**9,325**	**1.6%**	**1.4%**

Source: U.S. EPA analysis

Table 10-8: Estimated Water Quality Improvements Under Option 2

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		
	RFI Miles Receiving Construction Discharges	Miles of River in RF1 Network	River Miles	% of River Miles Receiving Construction Discharges	% of Total River Miles	River Miles	% of River Miles Receiving Construction Discharges	% of Total River Miles	River Miles	% of River Miles Receiving Construction Discharges	% of Total River Miles	River Miles	% of River Miles Receiving Construction Discharges	% of Total River Miles
1	16,952	18,293	1,510	8.9%	8.3%	38	0.2%	0.2%	0	0.0%	0.0%	1,548	9.1%	8.5%
2	15,888	16,110	5,345	33.6%	33.2%	196	1.2%	1.2%	7	0.0%	0.0%	5,548	34.9%	34.4%
3	29,183	33,617	16,244	55.7%	48.3%	2,356	8.1%	7.0%	244	0.8%	0.7%	18,844	64.6%	56.1%
4	87,312	94,464	30,169	34.6%	31.9%	13,243	15.2%	14.0%	4,817	5.5%	5.1%	48,229	55.2%	51.1%
5	71,508	71,550	7,156	10.0%	10.0%	990	1.4%	1.4%	138	0.2%	0.2%	8,283	11.6%	11.6%
6	82,855	98,669	5,734	6.9%	5.8%	6,850	8.3%	6.9%	3,924	4.7%	4.0%	16,507	19.9%	16.7%
7	54,018	60,909	2,310	4.3%	3.8%	1,004	1.9%	1.6%	477	0.9%	0.8%	3,791	7.0%	6.2%
8	113,710	130,311	278	0.2%	0.2%	0	0.0%	0.0%	9	0.0%	0.0%	288	0.3%	0.2%
9	47,474	56,259	342	0.7%	0.6%	46	0.1%	0.1%	0	0.0%	0.0%	388	0.8%	0.7%
10	66,468	69,495	4,545	6.8%	6.5%	2,370	3.6%	3.4%	1,076	1.6%	1.5%	7,991	12.0%	11.5%
National Total	**585,368**	**649,676**	**73,632**	**12.6%**	**11.3%**	**27,093**	**4.6%**	**4.2%**	**10,693**	**1.8%**	**1.6%**	**111,418**	**19.0%**	**17.1%**

Source: U.S. EPA analysis

Table 10-9: Estimated Water Quality Improvements Under Option 3

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		
	RFI Miles Receiving Construction Discharges	Miles of River in RF1 Network	River Miles	% of River Miles Receiving Construction Discharges	% of Total River Miles	River Miles	% of River Miles Receiving Construction Discharges	% of Total River Miles	River Miles	% of River Miles Receiving Construction Discharges	% of Total River Miles	River Miles	% of River Miles Receiving Construction Discharges	% of Total River Miles
1	16,952	18,293	2,754	16.2%	15.0%	192	1.1%	1.0%	9	0.1%	0.0%	2,956	17.4%	16.1%
2	15,888	16,110	6,864	43.2%	42.6%	621	3.9%	3.9%	30	0.2%	0.2%	7,514	47.3%	46.6%
3	29,183	33,617	16,843	57.7%	50.1%	3,631	12.4%	10.8%	355	1.2%	1.1%	20,830	71.4%	62.0%
4	87,312	94,464	26,844	30.7%	28.4%	19,397	22.2%	20.5%	8,486	9.7%	9.0%	54,727	62.7%	57.9%
5	71,508	71,550	9,310	13.0%	13.0%	1,883	2.6%	2.6%	414	0.6%	0.6%	11,606	16.2%	16.2%
6	82,855	98,669	5,212	6.3%	5.3%	7,095	8.6%	7.2%	5,395	6.5%	5.5%	17,702	21.4%	17.9%
7	54,018	60,909	2,973	5.5%	4.9%	1,041	1.9%	1.7%	735	1.4%	1.2%	4,749	8.8%	7.8%
8	113,710	130,311	3,603	3.2%	2.8%	362	0.3%	0.3%	128	0.1%	0.1%	4,092	3.6%	3.1%
9	47,474	56,259	1,786	3.8%	3.2%	306	0.6%	0.5%	101	0.2%	0.2%	2,193	4.6%	3.9%
10	66,468	69,495	4,489	6.8%	6.5%	3,367	5.1%	4.8%	2,261	3.4%	3.3%	10,118	15.2%	14.6%
National Total	**585,368**	**649,676**	**80,678**	**13.8%**	**12.4%**	**37,895**	**6.5%**	**5.8%**	**17,915**	**3.1%**	**2.8%**	**136,488**	**23.3%**	**21.0%**

Source: U.S. EPA analysis

10.2 Total Benefits of 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 E* 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 WTP based on assigned values for model variables, chosen to represent a resource change in the C&D policy context. EPA's meta-analysis results imply a simple benefit function of the following general form:

$$\ln(\text{WTP}) = \text{intercept} + \sum(\text{coefficient}_i)(\text{Independent Variable Values}_i) \quad \text{(Eq. 10-2)}$$

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 water bodies); 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 10-10* provides the estimated regression equation *intercept* (5.58), variable coefficients (*coefficient*$_i$), and the corresponding independent variable names. *Appendix E* 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 10-10* 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 9.49, 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 E* 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 water body 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 water body type variables (including *regional_fresh*) to zero because the default value is a resource change taking place in estuaries. [13]

Water quality improvements resulting from the proposed 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: Water Quality Index*. 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.

[13] The water quality model used in this analysis (SPARROW) does not go beyond the terminal reach to estimated water quality in large water bodies, and these water bodies 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 water bodies under the post-compliance scenario (see *Chapter 6* for detail).

Table 10-10: Independent Variable Assignments

Variable Type	Variable	Coefficient	Assigned Value	Explanation
Study and Methodology Variables	*intercept*	5.5802	1	Set to *one* as a default value.
	year_index	-0.0774	9.49	Set to 9.49 to reflect the mean year that the studies in the dataset were conducted.
	discrete	-0.1198	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	-1.3255	0	Set to *zero* because hypothetical voluntary payment mechanisms are not even potentially incentive compatible (Mitchell and Carson 1989)
	mail	-0.2198	0	Set to *zero* because mail surveys are believed to be of less quality than in-person interviews.
	lump_sum	0.5468	0	Set to *zero* because the policy option will be paid for over a period of years.
	WQI	-0.3238	1	Set to *one* because of the methodological use of the WQI in the meta-analysis.
	nonparam	-0.6732	0	Set to *zero* because most studies used in the meta analysis used regression analysis to calculate willingness to pay values
	non_reviewed	-0.2757	0	Set to *zero* because studies published in peer-reviewed journals are preferred.
	median_WTP	-0.514	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	-0.8736	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*	0.00000247	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	-0.4093	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).
Water Body Type	*single_river*	-0.3971	0	For valuing improvement in freshwater bodies, *regional_fresh* is set to *one* because
	single_lake	-0.0575	0	

Table 10-10: Independent Variable Assignments

Variable Type	Variable	Coefficient	Assigned Value	Explanation
Variables	*multiple_river*	-1.4554	0	the expectation is that multiple freshwater bodies within a state would be affected by the regulation. For saltwater reaches, all water body type variables are set to zero because the default value in the model is a resource change taking place in estuaries.
	regional_fresh	0.1773	Varies	
	salt_pond	1.0251	0	
Geographic Region and Scale Variables	*num_riv_ pond*	0.1155	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	-0.8784	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	1.5114	0	
Resource Improvement Variables	*allmult*	-0.3755	1	Set to *one* because it is assumed that multiple species would benefit from water quality improvements
	nonspec	-0.3849	0	Set to *zero* because the number of water bodies affected is known.
	lnquality_ch	0.4359	Varies	Set to the natural log of average change in the WQI for a given state.
	fish_use	-0.3478	1	Set to *one* because the regulation is expected to benefit a variety of aquatic species and therefore enhance recreational fishing opportunities.
	fishplus	0.4343	0	Set to *zero* because the regulation is not expected to result in a fish population change of 50 percent or greater.
	lnbase	0.0317	Varies	Set to the natural log of the average base water quality for a given state.

Source: U.S. EPA Analysis

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 corresponding to four levels of baseline water quality conditions ($WQI \leq 25$, $25 < WQI \leq 50$, $50 < WQI \leq 70$, and $70 < WQI$) and three different ranges of the expected water quality improvements ($\Delta WQI \leq 0.1$, $0.1 < \Delta WQI \leq 0.5$, $0.5 < \Delta WQI$). 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 10-10,* column 3) are multiplied by the variable levels chosen above *(Table 10-10,* 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 (0.1883) taken from *Table E-3* in *Appendix E*.

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 in their 1986 *Review of Economics and Statistics* paper "Approximating the Statistical Property of Elasticities." 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 lower or upper bound 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 confidence intervals of total WTP for each EPA region, based on the results of the total WTP regression model. *Table 10-11* 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 10-11 presents low, mid, and high estimates of household WTP for water quality improvements resulting form reduced sediment discharge from construction sites by EPA region and policy option.

Table 10-11: Average Household Willingness to Pay for Water Quality Improvement by Region (2008$)

EPA Region	Option 1			Option 2			Option 3		
	Lower 10% Bound	Mean	Upper 90% Bound	Lower 10% Bound	Mean	Upper 90% Bound	Lower 10% Bound	Mean	Upper 90% Bound
1	$0.00	$0.00	$0.00	$0.06	$0.31	$0.67	$0.36	$1.84	$3.67
2	$0.00	$0.02	$0.04	$0.65	$3.11	$6.27	$1.02	$4.61	$9.07
3	$0.00	$0.00	$0.00	$1.47	$5.90	$12.24	$1.66	$6.59	$13.68
4	$0.21	$0.70	$1.54	$2.06	$5.12	$11.03	$3.19	$7.09	$15.51
5	$0.00	$0.01	$0.03	$0.32	$1.28	$2.64	$0.46	$1.68	$3.63
6	$0.00	$0.00	$0.00	$1.43	$2.28	$5.91	$1.74	$2.44	$6.71
7	$0.00	$0.00	$0.00	$0.54	$1.03	$2.40	$0.72	$1.22	$2.97
8	$0.00	$0.00	$0.01	$0.01	$0.04	$0.08	$0.16	$0.57	$1.15
9	$0.00	$0.00	$0.00	$0.03	$0.11	$0.23	$0.16	$0.62	$1.25
10	$0.00	$0.00	$0.00	$0.91	$2.37	$4.65	$1.42	$3.36	$6.50
National	**$0.04**	**$0.15**	**$0.32**	**$0.92**	**$2.59**	**$5.58**	**$1.33**	**$3.50**	**$7.59**

Source: U.S. EPA analysis

As shown in *Table 10-11* the estimated national average household WTP for water quality improvements resulting from the regulation range from $0.15 to $3.50 per household. The estimated WTP values vary greatly across EPA regions depending on the policy option and the level of construction activity in a given region.

EPA estimates that Option 1 is not expected to result in significant water quality improvements across all EPA regions. Region 4 is the only geographic region that shows any significant water quality improvements. The estimated WTP per household in Region 4 has a 10 percent lower bound of $0.21 per household and a 90 percent upper bound of $1.54 per household, with an average WTP of $0.70. The estimated WTP for water quality improvements in Regions 1, 3, 6, 7, 9, and 10 is negligible. Nationwide, household WTP has a mean value of $0.15 and 90 percent confidence interval bounds ranging from $0.04 to $0.32.

The proposed policy option (Option 2) is estimated to improve ambient water quality in 19.0 percent of RF1 river miles included in the analysis. The estimated national average WTP for water quality improvements resulting from the regulation is $2.59 per household per year. Regions 3 and 4 are estimated to see improvements in water quality in more than 40 percent of their total RF1 river miles (*Section 10.1.5*). EPA's analysis suggests that Region 3 households would be willing to pay the most ($5.90 per household per year) for water quality improvements resulting from the C&D regulation. Region 4 has the second largest household WTP of $5.12. Conversely, households located in Region 8 are estimated to have the lowest WTP for water quality improvements from the regulation, $0.04.

Policy Option 3 yields the most significant results overall in terms of RF1 miles expected to improve under the post-compliance scenario. The estimated scale of improvements ranges from 3.6 percent to 71.4 percent of total RF1 river miles in Region 8 and 3, respectively (*Section 10.1.5*). Nationwide, the 90 percent confidence interval for the estimated per-household WTP has a 10 percent lower bound of $1.33 and a 90 percent upper bound of $7.59, with a mean value of $3.50.

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 river 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 (Eq. 10-4):

$$TWTP_{reach} = WTP(WQI_{baseline}, \Delta WQI) \times StateHH \times PercentRiverMiles \quad \text{(Eq. 10-4)}$$

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 river 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 10-12* presents estimated benefits of the regulation by EPA region and policy option.

Table 10-12 :Regional Willingness to Pay for Water Quality Improvement (Millions 2008$)

EPA Region	Option 1			Option 2			Option 3		
	Lower 10% Bound	Mean	Upper 90% Bound	Lower 10% Bound	Mean	Upper 90% Bound	Lower 10% Bound	Mean	Upper 90% Bound
1	$0.00	$0.00	$0.00	$0.32	$1.76	$3.77	$2.04	$10.44	$20.76
2	$0.02	$0.18	$0.39	$6.81	$32.66	$65.72	$10.71	$48.32	$95.11
3	$0.00	$0.00	$0.00	$16.92	$67.86	$140.83	$19.14	$75.90	$157.42
4	$4.86	$16.10	$35.60	$47.57	$118.34	$254.72	$73.77	$163.74	$358.19
5	$0.04	$0.28	$0.61	$6.53	$25.98	$53.67	$9.37	$34.09	$73.78
6	$0.00	$0.00	$0.00	$18.89	$30.08	$77.78	$22.96	$32.16	$88.40
7	$0.00	$0.00	$0.00	$2.96	$5.62	$13.11	$3.92	$6.67	$16.22
8	<$0.00	$0.02	$0.04	$0.05	$0.17	$0.33	$0.64	$2.25	$4.53
9	$0.00	$0.00	$0.00	$0.39	$1.74	$3.59	$2.49	$9.63	$19.46
10	$0.00	$0.00	$0.00	$4.16	$10.82	$21.19	$6.49	$15.32	$29.62
National	**$4.93**	**$16.57**	**$36.63**	**$104.60**	**$295.01**	**$634.72**	**$151.53**	**$398.53**	**$863.49**

Source: U.S. EPA analysis

EPA estimates that total annual benefits of water quality improvements resulting from reduced sediment discharge from construction sites range from $16.57 million under Option 1 to $398.53 million under Option 3. The estimated mean regional benefits vary from $0.00 to $163.74 million per year, depending on the level of construction activity and average rainfall in a given region and stringency of the policy option.

As shown in *Table 10-12*, Option 1 is not expected to result in significant water quality improvements in most EPA regions. Thus, this option yields the smallest benefits at the regional and national levels. The national benefits of water quality improvements under this option have a 10 percent lower bound estimate of $4.93 million, a 90 percent upper bound estimate of $36.63 million, and a national average of $16.57 million. Region 4 gains the most benefit from water quality improvement, with a total value of $16.10 million (97 percent of the total national benefits). Region 5 has the second largest benefits ($0.28 million), which account for 1.7 percent of the total national benefits.

Under Option 2, the average national benefits are $295.01 million. The 90 percent confidence interval has a 10 percent lower bound of $104.60 million and a 90 percent upper bound value of $634.72 million. Region 4 gains the most from water quality improvements resulting from the regulation ($118.34 million). EPA Region 8 receives the smallest benefits, $0.17 million.

Under Option 3, the estimated mean national benefits of water quality improvement from the regulation are $398.53 million with a 10 percent lower bound of $151.53 and a 90 percent upper bound of $863.49 million. As with the other policy options, Region 4 receives the largest benefits from water quality improvements, accounting for 41 percent ($163.74 million) of the total national benefits. Region 8 is anticipated to gain least under this Option 3, with the total regional benefits estimated at $2.25 million per year.

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 (Rosenberger and Stanley 2006; Rosenberger and Phipps 2007; Smith et al. 2002). 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. 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. 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.

- 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. 2006; 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 C* 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 associated with the use of SPARROW are discussed in *Chapter 11* of this report.

11 Total Estimated Benefits

This chapter summarizes findings from EPA's analysis of the expected benefits from the proposed 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: Sediment basins of a certain capacity required on all sites larger than 10 acres draining to one location; construction general permit for other sites.
- Option 2 (in addition to the requirements of Option 1): Turbidity standard for sites with 30 or more acres *and* located in areas with an "R-factor"[14] greater than 50 *and* that have soils that contain more than 10 percent (by mass) particles less than 2 microns in diameter. Sites larger than 30 acres not meeting both of these conditions may obtain a waiver from the turbidity standard.
- Option 3 (in addition to the requirements of Option 1): Turbidity standard for all sites larger than 10 acres (no waivers available).

Chapter 5: Pollutant Load Modeling 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 E* 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 proposed regulation.

EPA was unable to quantify benefits stemming from:

- Enhanced commercial fishing
- Industrial and agricultural use of water resources
- Reduced risk of flooding, and an increase in property values.

In addition, EPA's analysis of monetized benefits is limited to reaches included in the RF1 network. Therefore, the monetized benefits presented in this section are underestimates because they (1) exclude several benefit categories and (2) do not reflect benefits that will be incurred by complex water bodies (i.e., estuaries and Great Lakes), or by streams outside of the RF1 network.

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,* and *Table 11-4* 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

[14] The "R-factor" refers to the rainfall-runoff erosivity factor in RUSLE, which represents the amount of rainfall and its potential to contribute to soil erosion based on rainfall patterns. More detail is available in *Chapter 5* and in the Technical Development Document (EPA 2008b) for this regulation.

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 between 6 and 20 percent of total benefits (depending on estimate range and regulatory option), varying it has little effect on the total benefits estimate. The remaining discussion presents the benefits estimates assuming a 3 percent discount rate; the associated tables present results for both discount rates. EPA calculated benefits for drinking water treatment and WTP using a single-year timeframe, which did not require discounting or annualizing. All benefits presented reflect annual values.

Total national benefits vary significantly among the three regulatory options: under Option 1, the estimated benefits range from approximately $6.4 million to approximately $39.1 million, with a midpoint estimate of $18.4 million. Estimated avoided costs range from $1.4 million to $2.5 million, with a midpoint of $1.8 million, and WTP varies between $4.9 and $36.6 million at the 5 and 95 percent confidence intervals, with a mean estimate of $16.6 million.

As shown in *Table 11-1*, under Option 2, the estimated benefits range from $135.9 million to $683.5 million, with a midpoint estimate of $332.9 million. Nonmarket benefits estimated based on household WTP for surface water quality improvements account for 77, 89, and 93 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 2 ranges from $104.6 to $634.7 million, with a mean value of $295.0 million. The estimated cost savings to industry and government through reduced costs of navigable waterway maintenance, reservoir dredging, and drinking water treatment range from $31.3 million to $48.8 million per year, with a midpoint estimate of $37.9 million. Under Option 2, avoided cost benefits account for 23, 11, and 7 percent of total benefits in the low, mid, and high estimates, respectively.

Under Option 3, total benefits are estimated to be between $209.6 and $956.1 million, with a midpoint estimate of $469.5 million. The avoided costs are estimated to be between $58.0 and $92.6 million per year, with a midpoint estimate of $70.9 million. WTP under Option 3 ranges between $151.5 million and $863.5 million at the 5 and 95 percent confidence intervals, with a mean value of $398.5 million.

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	$0.7	$1.0	$1.6	$0.7	$0.9	$1.8
Water Storage[1]	$0.6	$0.6	$0.6	$0.5	$0.5	$0.6
Drinking Water[1]	$0.2	$0.2	$0.2	$0.2	$0.2	$0.2
Avoided Costs	*$1.4*	*$1.8*	*$2.5*	*$1.3*	*$1.7*	*$2.6*
WTP[1]	$4.9	$16.6	$36.6	$4.9	$16.6	$36.6
Total[2]	**$6.4**	**$18.4**	**$39.1**	**$6.3**	**$18.3**	**$39.2**
Option 2						
Navigation	$8.9	$12.9	$22.1	$8.8	$12.6	$23.8
Water Storage[1]	$16.5	$17.6	$18.7	$13.7	$15.9	$18.3
Drinking Water[1]	$5.8	$7.4	$8.0	$5.8	$7.4	$8.0
Avoided Costs	*$31.3*	*$37.9*	*$48.8*	*$28.3*	*$35.9*	*$50.1*
WTP[1]	$104.6	$295.0	$634.7	$104.6	$295.0	$634.7
Total[2]	**$135.9**	**$332.9**	**$683.5**	**$132.9**	**$330.9**	**$684.8**

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 3						
Navigation	$18.9	$27.2	$45.7	$18.7	$26.5	$49.2
Water Storage[1]	$28.7	$30.6	$32.5	$23.8	$27.6	$31.8
Drinking Water[1]	$10.4	$13.1	$14.5	$10.4	$13.1	$14.5
Avoided Costs	*$58.0*	*$70.9*	*$92.6*	*$52.9*	*$67.3*	*$95.5*
WTP[1]	$151.5	$398.5	$863.5	$151.5	$398.5	$863.5
Total[2]	**$209.6**	**$469.5**	**$956.1**	**$204.4**	**$465.8**	**$959.0**

[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.
Source: USEPA analysis

Table 11-2, *Table 11-3*, and *Table 11-4* 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 river miles and the most significant reductions in sediment concentrations in these river miles. Together, the magnitude and extent of these changes produce larger cost savings and water quality improvements in this region than in any other.

Under Policy Option 1, a construction general permit for all sites and sediment basin requirements for large sites, total benefits range between approximately $6.4 million in the low estimate and $39.1 million in the high estimate, with a midpoint estimate of $18.4 million. Region 4 is the sole region with expected benefits totaling more than $1 million, with estimates ranging from $6.2 to $38.0 million. The estimated benefits for other regions are modest at best, with midpoint estimates ranging from less than $500 in Region 10 to $291,000 in Region 5.

Policy Option 2, which adds a turbidity standard to the regulation for construction sites meeting certain conditions, is expected to result in significantly larger benefits in all regions. EPA expects total benefits under this option to range between $135.9 and $683.5 million, with a midpoint estimate of $332.9 million. Under this regulatory option, Regions 2, 3, 4, 5, 6, and 10 benefit most substantially, with midpoint estimates exceeding $25 million in each of Regions 2 through 6. Region 4 alone accounts for 41 to 48 percent of the total national benefits anticipated under this option, ranging from $65.4 million to $284.3 million, with a midpoint estimate of $140.0 million. In Region 3, total benefits are estimated to range between $18.5 million and $143.2 million, with a midpoint estimate of $69.7 million. In Regions 2, 5, 6, and 10, midpoint benefits estimates range between $13.4 and $38.0 million. The Agency expects benefits for this option to be substantially larger than those under Option 1, because a turbidity standard targeting stormwater discharge from construction sites is likely to produce higher benefits in terms of TSS and turbidity reductions. However, areas of the country with low rainfall or low percentage of clay in their soils, such as Regions 8 and 9, do not benefit as significantly because many sites in these areas are likely to qualify for waivers of the turbidity standard. Conversely, construction sites are not likely to affect sediment or turbidity pollution on a large scale in these regions.

Option 3, under which all large sites are required to meet an effluent turbidity standard, increases the midpoint estimate of total national benefits by 41 percent compared to Option 2. This option is expected to generate benefits of $469.5 million, with a range of $209.6 to $956.1 million between the low and high estimates. Regions 2, 3, 4, 5, 6, and 10 are again estimated to benefit most under this option, though Option 3 is also estimated to substantially increase benefits in Regions 8 and 9, where many areas are

exempt from the turbidity standard under Option 2. The Agency's midpoint estimates for benefits in these regions under Option 3 are $2.6 million and $10.2 million, respectively.

Table 11-2: Annual Total National Benefits Under Option 1 (thousands of 2008$)

EPA Region	Benefits to Navigation Discounted at 3%			Benefits to Navigation Discounted at 7%		
	Low	Mid	High	Low	Mid	High
1	$1	$1	$3	$1	$1	$3
2	$31	$183	$396	$31	$183	$396
3	$2	$3	$4	$2	$3	$4
4	$6,273	$17,851	$38,044	$6,169	$17,769	$38,157
5	$52	$291	$626	$51	$290	$626
6	$2	$2	$4	$2	$2	$4
7	$1	$1	$1	$0	$1	$1
8	$7	$23	$46	$6	$23	$46
9	$1	$1	$1	$1	$1	$1
10[1]	$0	$0	$0	$0	$0	$0
Total[2]	**$6,370**	**$18,357**	**$39,124**	**$6,263**	**$18,273**	**$39,239**

[1] Benefits estimates in this region are not zero, but less than $500 annually.
[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.
Source: USEPA analysis

Table 11-3: Annual Total National Benefits Under Option 2 (thousands of 2008$)

EPA Region	Benefits to Navigation Discounted at 3%			Benefits to Navigation Discounted at 7%		
	Low	Mid	High	Low	Mid	High
1	$425	$1,877	$3,932	$410	$1,867	$3,935
2	$7,230	$33,322	$66,814	$7,214	$33,302	$66,875
3	$18,463	$69,720	$143,177	$18,377	$69,654	$143,270
4	$65,399	$140,044	$284,250	$63,790	$138,884	$285,328
5	$7,421	$27,010	$54,847	$7,319	$26,946	$54,850
6	$25,659	$37,976	$87,051	$24,916	$37,501	$87,120
7	$4,745	$7,598	$15,247	$4,496	$7,448	$15,217
8	$88	$209	$373	$82	$205	$372
9	$440	$1,790	$3,661	$436	$1,788	$3,662
10	$6,007	$13,357	$24,178	$5,906	$13,292	$24,186
Total[1]	**$135,877**	**$332,903**	**$683,530**	**$132,945**	**$330,887**	**$684,817**

[1] Totals may not be equal to the sum of regional data because the WTP model estimates the national total rather than summing regional totals.
Source: USEPA analysis

Table 11-4: Annual Total National Benefits Under Option 3 (thousands of 2008$)

EPA Region	Benefits to Navigation Discounted at 3%			Benefits to Navigation Discounted at 7%		
	Low	Mid	High	Low	Mid	High
1	$2,315	$10,758	$21,178	$2,282	$10,737	$21,187
2	$11,442	$49,440	$96,897	$11,415	$49,407	$96,991
3	$21,410	$78,646	$160,880	$21,285	$78,549	$161,020
4	$109,071	$207,357	$417,892	$106,228	$205,203	$420,483
5	$10,812	$35,759	$75,696	$10,652	$35,657	$75,705
6	$33,319	$44,255	$102,551	$32,187	$43,532	$102,655
7	$6,415	$9,444	$19,215	$6,066	$9,234	$19,173
8	$924	$2,560	$4,852	$881	$2,534	$4,846
9	$2,968	$10,187	$20,143	$2,923	$10,158	$20,153
10	$10,901	$21,048	$36,769	$10,507	$20,797	$36,797
Total[1]	**$209,577**	**$469,455**	**$956,073**	**$204,425**	**$465,809**	**$959,010**

[1] Totals may not be equal to the sum of regional data because the WTP model estimates the national total rather than summing regional totals.

Source: USEPA analysis

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 benefit 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 water bodies, coverage is limited in certain important respects. First, RF1 network coverage is limited to the conterminous 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, at 1:24,000 - 1:100,000 scale, is currently over 7 million

miles (USGS 2007). Given that RF1 accounts only for 10 percent of the total river 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.

- Omission of coastal waters from the analysis of monetized benefits. The SPARROW model does not allow prediction of water quality changes in coastal waters. Therefore, all coastal waters in the United States (10,630 shoreline miles) are omitted from the analysis of monetized benefits.[15] Because the estimated willingness to pay for water quality improvements is a function of the total river and shoreline miles that are expected to improve from reduced sediment discharges, this omission is likely to lead to understatement of the estimated willingness to pay for water quality improvements resulting from the proposed regulation.
- 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 2007d) reports 40.6 million acres of lakes and reservoirs in the conterminous 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 2007d).[16] Omission of these water body 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 One Pollutant of Concern (Sediment)

Existing case studies of environmental impacts associated with construction activities demonstrated that a number of pollutants are found in construction site discharges, including turbidity, nitrogen, phosphorus, BOD, metals, toxic organics, trash and debris, and other miscellaneous pollutants. However, EPA's analysis of benefits from reduced construction site discharges focuses on total suspended solids only. This is likely to result in underestimation of the expected water quality changes resulting from the proposed

[15] EPA was able to estimate ambient concentrations of sediment (Total Suspended Solids (TSS)) within estuaries using SPARROW output in conjunction with the Dissolved Concentration Potential (DCP) approach (Versar, Inc./USEPA 1997). However, estuarine reaches were not included in estimation of monetized benefits due to time constraints.

[16] The estimated total lake/reservoir acres do not include the Great Lakes.

regulation because the combined impact of several pollutants on ambient water quality conditions is likely to be greater than a single pollutant's impact.

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 uncertain, they may not be trivial. *Chapter 4* 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 water bodies. Reducing sediment discharges from construction sites is likely to increase market values of properties located in the vicinity of 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).[17]
- 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 proposed regulation is expected to reduce the costs of ditch maintenance by reducing the amount of sediment deposited in ditches.
- Industrial water use. The proposed 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 proposed 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.

[17] The nonmarket component (i.e., increased satisfaction with the property) is implicitly accounted for in WTP for improvements in environmental services provided by surface waters affected by construction site discharges (see *Chapter 10* for detail).

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Appendix A – Threatened and Endangered Aquatic Species

Table A-1: List of Federal Threatened and Endangered Aquatic Species Potentially Impacted by Sediment

Common name	Scientific Name	Threatened/Endangered Status
Fish		
Catfish, Yaqui	*Ictalurus pricei*	Threatened
Cavefish, Alabama	*Speoplatyrhinus poulsoni*	Endangered
Cavefish, Ozark	*Amblyopsis rosae*	Threatened
Chub, bonytail	*Gila elegans*	Endangered
Chub, Borax Lake	*Gila boraxobius*	Endangered
Chub, Chihuahua	*Gila nigrescens*	Threatened
Chub, Gila	*Gila intermedia*	Endangered
Chub, humpback	*Gila cypha*	Endangered
Chub, Hutton tui	*Gila bicolor ssp.*	Threatened
Chub, Mohave tui	*Gila bicolor mohavensis*	Endangered
Chub, Oregon	*Oregonichthys crameri*	Endangered
Chub, Owens tui	*Gila bicolor snyderi*	Endangered
Chub, Pahranagat roundtail	*Gila robusta jordani*	Endangered
Chub, slender	*Erimystax cahni*	Threatened
Chub, Sonora	*Gila ditaenia*	Threatened
Chub, spotfin	*Erimonax monachus*	Threatened
Chub, Virgin River	*Gila seminuda*	Endangered
Chub, Yaqui	*Gila purpurea*	Endangered
Cui-ui	*Chasmistes cujus*	Endangered
Dace, Ash Meadows speckled	*Rhinichthys osculus nevadensis*	Endangered
Dace, blackside	*Phoxinus cumberlandensis*	Threatened
Dace, Clover Valley speckled	*Rhinichthys osculus oligoporus*	Endangered
Dace, desert	*Eremichthys acros*	Threatened
Dace, Foskett speckled	*Rhinichthys osculus ssp.*	Threatened
Dace, Independence Valley speckled	*Rhinichthys osculus lethoporus*	Endangered
Dace, Kendall Warm Springs	*Rhinichthys osculus thermalis*	Endangered
Dace, Moapa	*Moapa coriacea*	Endangered
Darter, amber	*Percina antesella*	Endangered
Darter, bayou	*Etheostoma rubrum*	Threatened
Darter, bluemask (=jewel)	*Etheostoma sp.*	Endangered
Darter, boulder	*Etheostoma wapiti*	Endangered
Darter, Cherokee	*Etheostoma scotti*	Threatened
Darter, duskytail	*Etheostoma percnurum*	Endangered
Darter, Etowah	*Etheostoma etowahae*	Endangered
Darter, fountain	*Etheostoma fonticola*	Endangered
Darter, goldline	*Percina aurolineata*	Threatened
Darter, leopard	*Percina pantherina*	Threatened
Darter, Maryland	*Etheostoma sellare*	Endangered
Darter, Niangua	*Etheostoma nianguae*	Threatened
Darter, Okaloosa	*Etheostoma okaloosae*	Endangered
Darter, relict	*Etheostoma chienense*	Endangered

Table A-1: List of Federal Threatened and Endangered Aquatic Species Potentially Impacted by Sediment

Common name	Scientific Name	Threatened/Endangered Status
Darter, slackwater	*Etheostoma boschungi*	Threatened
Darter, snail	*Percina tanasi*	Threatened
Darter, vermilion	*Etheostoma chermocki*	Endangered
Darter, watercress	*Etheostoma nuchale*	Endangered
Gambusia, Big Bend	*Gambusia gaigei*	Endangered
Gambusia, Clear Creek	*Gambusia heterochir*	Endangered
Gambusia, Pecos	*Gambusia nobilis*	Endangered
Gambusia, San Marcos	*Gambusia georgei*	Endangered
Goby, tidewater	*Eucyclogobius newberryi*	Endangered
Logperch, Conasauga	*Percina jenkinsi*	Endangered
Logperch, Roanoke	*Percina rex*	Endangered
Madtom, Neosho	*Noturus placidus*	Threatened
Madtom, pygmy	*Noturus stanauli*	Endangered
Madtom, Scioto	*Noturus trautmani*	Endangered
Madtom, smoky	*Noturus baileyi*	Endangered
Madtom, yellowfin	*Noturus flavipinnis*	Threatened
Minnow, Devils River	*Dionda diaboli*	Threatened
Minnow, loach	*Tiaroga cobitis*	Threatened
Minnow, Rio Grande silvery	*Hybognathus amarus*	Endangered
Pikeminnow, Colorado	*Ptychocheilus lucius*	Endangered
Poolfish, Pahrump	*Empetrichthys latos*	Endangered
Pupfish, Ash Meadows Amargosa	*Cyprinodon nevadensis mionectes*	Endangered
Pupfish, Comanche Springs	*Cyprinodon elegans*	Endangered
Pupfish, desert	*Cyprinodon macularius*	Endangered
Pupfish, Devils Hole	*Cyprinodon diabolis*	Endangered
Pupfish, Leon Springs	*Cyprinodon bovinus*	Endangered
Pupfish, Owens	*Cyprinodon radiosus*	Endangered
Pupfish, Warm Springs	*Cyprinodon nevadensis pectoralis*	Endangered
Salmon, Atlantic	*Salmo salar*	Endangered
Salmon, chinook	*Oncorhynchus tshawytscha*	Threatened/Endangered
Salmon, chum	*Oncorhynchus keta*	Threatened
Salmon, coho	*Oncorhynchus kisutch*	Threatened/Endangered
Salmon, sockeye	*Oncorhynchus nerka*	Threatened/Endangered
Sawfish, smalltooth	*Pristis pectinata*	Endangered
Sculpin, pygmy	*Cottus paulus*	Threatened
Shiner, Arkansas River	*Notropis girardi*	Threatened
Shiner, beautiful	*Cyprinella formosa*	Threatened
Shiner, blue	*Cyprinella caerulea*	Threatened
Shiner, Cahaba	*Notropis cahabae*	Endangered
Shiner, Cape Fear	*Notropis mekistocholas*	Endangered
Shiner, palezone	*Notropis albizonatus*	Endangered
Shiner, Pecos bluntnose	*Notropis simus pecosensis*	Threatened
Shiner, Topeka	*Notropis topeka*	Endangered
Silverside, Waccamaw	*Menidia extensa*	Threatened
Smelt, delta	*Hypomesus transpacificus*	Threatened
Spikedace	*Meda fulgida*	Threatened

Table A-1: List of Federal Threatened and Endangered Aquatic Species Potentially Impacted by Sediment

Common name	Scientific Name	Threatened/Endangered Status
Spinedace, Big Spring	*Lepidomeda mollispinis pratensis*	Threatened
Spinedace, Little Colorado	*Lepidomeda vittata*	Threatened
Spinedace, White River	*Lepidomeda albivallis*	Endangered
Springfish, Hiko White River	*Crenichthys baileyi grandis*	Endangered
Springfish, Railroad Valley	*Crenichthys nevadae*	Threatened
Springfish, White River	*Crenichthys baileyi baileyi*	Endangered
Steelhead	*Oncorhynchus mykiss*	Threatened/Endangered
Stickleback, unarmored threespine	*Gasterosteus aculeatus williamsoni*	Endangered
Sturgeon, Alabama	*Scaphirhynchus suttkusi*	Endangered
Sturgeon, gulf	*Acipenser oxyrinchus desotoi*	Threatened
Sturgeon, North American green	*Acipenser medirostris*	Threatened
Sturgeon, pallid	*Scaphirhynchus albus*	Endangered
Sturgeon, shortnose	*Acipenser brevirostrum*	Endangered
Sturgeon, white	*Acipenser transmontanus*	Endangered
Sucker, June	*Chasmistes liorus*	Endangered
Sucker, Lost River	*Deltistes luxatus*	Endangered
Sucker, Modoc	*Catostomus microps*	Endangered
Sucker, razorback	*Xyrauchen texanus*	Endangered
Sucker, Santa Ana	*Catostomus santaanae*	Threatened
Sucker, shortnose	*Chasmistes brevirostris*	Endangered
Sucker, Warner	*Catostomus warnerensis*	Threatened
Topminnow, Gila (incl. Yaqui)	*Poeciliopsis occidentalis*	Endangered
Trout, Apache	*Oncorhynchus apache*	Threatened
Trout, bull	*Salvelinus confluentus*	Threatened
Trout, Gila	*Oncorhynchus gilae*	Threatened
Trout, greenback cutthroat	*Oncorhynchus clarki stomias*	Threatened
Trout, Lahontan cutthroat	*Oncorhynchus clarki henshawi*	Threatened
Trout, Little Kern golden	*Oncorhynchus aguabonita whitei*	Threatened
Trout, Paiute cutthroat	*Oncorhynchus clarki seleniris*	Threatened
Woundfin	*Plagopterus argentissimus*	Endangered
Amphibians		
Coqui, golden	*Eleutherodactylus jasperi*	Threatened
Frog, California red-legged	*Rana aurora draytonii*	Threatened
Frog, Chiricahua leopard	*Rana chiricahuensis*	Threatened
Frog, Mississippi gopher	*Rana capito sevosa*	Endangered
Frog, mountain yellow-legged	*Rana muscosa*	Endangered
Guajon	*Eleutherodactylus cooki*	Threatened
Salamander, Barton Springs	*Eurycea sosorum*	Endangered
Salamander, California tiger	*Ambystoma californiense*	Threatened/Endangered
Salamander, Cheat Mountain	*Plethodon nettingi*	Threatened
Salamander, desert slender	*Batrachoseps aridus*	Endangered
Salamander, flatwoods	*Ambystoma cingulatum*	Threatened
Salamander, Red Hills	*Phaeognathus hubrichti*	Threatened
Salamander, San Marcos	*Eurycea nana*	Threatened
Salamander, Santa Cruz long-toed	*Ambystoma macrodactylum croceum*	Endangered
Salamander, Shenandoah	*Plethodon shenandoah*	Endangered

Table A-1: List of Federal Threatened and Endangered Aquatic Species Potentially Impacted by Sediment

Common name	Scientific Name	Threatened/Endangered Status
Salamander, Sonora tiger	*Ambystoma tigrinum stebbinsi*	Endangered
Salamander, Texas blind	*Typhlomolge rathbuni*	Endangered
Toad, arroyo (=arroyo southwestern)	*Bufo californicus (=microscaphus)*	Endangered
Toad, Houston	*Bufo houstonensis*	Endangered
Toad, Puerto Rican crested	*Peltophryne lemur*	Threatened
Toad, Wyoming	*Bufo baxteri (=hemiophrys)*	Endangered
Mollusks		
Acornshell, southern	*Epioblasma othcaloogensis*	Endangered
Bankclimber, purple (mussel)	*Elliptoideus sloatianus*	Threatened
Bean, Cumberland (pearlymussel)	*Villosa trabalis*	Endangered
Bean, purple	*Villosa perpurpurea*	Endangered
Blossom, green (pearlymussel)	*Epioblasma torulosa gubernaculum*	Endangered
Blossom, tubercled (pearlymussel)	*Epioblasma torulosa torulosa*	Endangered
Blossom, turgid (pearlymussel)	*Epioblasma turgidula*	Endangered
Blossom, yellow (pearlymussel)	*Epioblasma florentina florentina*	Endangered
Catspaw (=purple cat's paw pearlymussel)	*Epioblasma obliquata obliquata*	Endangered
Catspaw, white (pearlymussel)	*Epioblasma obliquata perobliqua*	Endangered
Clubshell	*Pleurobema clava*	Endangered
Clubshell, black	*Pleurobema curtum*	Endangered
Clubshell, ovate	*Pleurobema perovatum*	Endangered
Clubshell, southern	*Pleurobema decisum*	Endangered
Combshell, Cumberlandian	*Epioblasma brevidens*	Endangered
Combshell, southern	*Epioblasma penita*	Endangered
Combshell, upland	*Epioblasma metastriata*	Endangered
Elktoe, Appalachian	*Alasmidonta raveneliana*	Endangered
Elktoe, Cumberland	*Alasmidonta atropurpurea*	Endangered
Fanshell	*Cyprogenia stegaria*	Endangered
Fatmucket, Arkansas	*Lampsilis powelli*	Threatened
Heelsplitter, Alabama (=inflated)	*Potamilus inflatus*	Threatened
Heelsplitter, Carolina	*Lasmigona decorata*	Endangered
Higgins eye (pearlymussel)	*Lampsilis higginsii*	Endangered
Kidneyshell, triangular	*Ptychobranchus greenii*	Endangered
Lampmussel, Alabama	*Lampsilis virescens*	Endangered
Lilliput, pale (pearlymussel)	*Toxolasma cylindrellus*	Endangered
Mapleleaf, winged	*Quadrula fragosa*	Endangered
Moccasinshell, Alabama	*Medionidus acutissimus*	Threatened
Moccasinshell, Coosa	*Medionidus parvulus*	Endangered
Moccasinshell, Gulf	*Medionidus penicillatus*	Endangered
Moccasinshell, Ochlockonee	*Medionidus simpsonianus*	Endangered
Monkeyface, Appalachian (pearlymussel)	*Quadrula sparsa*	Endangered
Monkeyface, Cumberland (pearlymussel)	*Quadrula intermedia*	Endangered
Mucket, orangenacre	*Lampsilis perovalis*	Threatened
Mucket, pink (pearlymussel)	*Lampsilis abrupta*	Endangered

Table A-1: List of Federal Threatened and Endangered Aquatic Species Potentially Impacted by Sediment

Common name	Scientific Name	Threatened/Endangered Status
Mussel, oyster	*Epioblasma capsaeformis*	Endangered
Mussel, scaleshell	*Leptodea leptodon*	Endangered
Pearlshell, Louisiana	*Margaritifera hembeli*	Threatened
Pearlymussel, birdwing	*Conradilla caelata*	Endangered
Pearlymussel, cracking	*Hemistena lata*	Endangered
Pearlymussel, Curtis	*Epioblasma florentina curtisii*	Endangered
Pearlymussel, dromedary	*Dromus dromas*	Endangered
Pearlymussel, littlewing	*Pegias fabula*	Endangered
Pigtoe, Cumberland	*Pleurobema gibberum*	Endangered
Pigtoe, dark	*Pleurobema furvum*	Endangered
Pigtoe, finerayed	*Fusconaia cuneolus*	Endangered
Pigtoe, flat	*Pleurobema marshalli*	Endangered
Pigtoe, heavy	*Pleurobema taitianum*	Endangered
Pigtoe, oval	*Pleurobema pyriforme*	Endangered
Pigtoe, rough	*Pleurobema plenum*	Endangered
Pigtoe, shiny	*Fusconaia cor*	Endangered
Pigtoe, southern	*Pleurobema georgianum*	Endangered
Pimpleback, orangefoot (pearlymussel)	*Plethobasus cooperianus*	Endangered
Pocketbook, fat	*Potamilus capax*	Endangered
Pocketbook, finelined	*Lampsilis altilis*	Threatened
Pocketbook, Ouachita rock	*Arkansia wheeleri*	Endangered
Pocketbook, shinyrayed	*Lampsilis subangulata*	Endangered
Pocketbook, speckled	*Lampsilis streckeri*	Endangered
Rabbitsfoot, rough	*Quadrula cylindrica strigillata*	Endangered
Riffleshell, northern	*Epioblasma torulosa rangiana*	Endangered
Riffleshell, tan	*Epioblasma florentina walkeri (=E. walkeri)*	Endangered
Ring pink (mussel)	*Obovaria retusa*	Endangered
Slabshell, Chipola	*Elliptio chipolaensis*	Threatened
Spinymussel, James	*Pleurobema collina*	Endangered
Spinymussel, Tar River	*Elliptio steinstansana*	Endangered
Stirrupshell	*Quadrula stapes*	Endangered
Three-ridge, fat (mussel)	*Amblema neislerii*	Endangered
Wartyback, white (pearlymussel)	*Plethobasus cicatricosus*	Endangered
Wedgemussel, dwarf	*Alasmidonta heterodon*	Endangered
Crustaceans		
Amphipod, Hay's Spring	*Stygobromus hayi*	Endangered
Amphipod, Illinois cave	*Gammarus acherondytes*	Endangered
Amphipod, Kauai cave	*Spelaeorchestia koloana*	Endangered
Amphipod, Noel's	*Gammarus desperatus*	Endangered
Amphipod, Peck's cave	*Stygobromus (=Stygonectes) pecki*	Endangered
Crayfish, cave	*Cambarus aculabrum*	Endangered
Crayfish, cave	*Cambarus zophonastes*	Endangered
Crayfish, Nashville	*Orconectes shoupi*	Endangered
Crayfish, Shasta	*Pacifastacus fortis*	Endangered

Table A-1: List of Federal Threatened and Endangered Aquatic Species Potentially Impacted by Sediment

Common name	Scientific Name	Threatened/Endangered Status
Fairy shrimp, Conservancy	*Branchinecta conservatio*	Endangered
Fairy shrimp, longhorn	*Branchinecta longiantenna*	Endangered
Fairy shrimp, Riverside	*Streptocephalus woottoni*	Endangered
Fairy shrimp, San Diego	*Branchinecta sandiegonensis*	Endangered
Fairy shrimp, vernal pool	*Branchinecta lynchi*	Threatened
Isopod, Lee County cave	*Lirceus usdagalun*	Endangered
Isopod, Madison Cave	*Antrolana lira*	Threatened
Isopod, Socorro	*Thermosphaeroma thermophilus*	Endangered
Shrimp, Alabama cave	*Palaemonias alabamae*	Endangered
Shrimp, California freshwater	*Syncaris pacifica*	Endangered
Shrimp, Kentucky cave	*Palaemonias ganteri*	Endangered
Shrimp, Squirrel Chimney Cave	*Palaemonetes cummingi*	Threatened
Tadpole shrimp, vernal pool	*Lepidurus packardi*	Endangered
Corals		
Coral, elkhorn	*Acropora palmata*	Threatened
Coral, staghorn	*Acropora cervicornis*	Threatened

Appendix B – SPARROW Model Documentation

B.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.

B.2 Modeling Concept

B.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 proposed 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 (BMP) 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.

B.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; Alexander et al., 2002a; McMahon et al., 2003), nutrient and suspended sediment removal rates in reservoirs (Alexander et al., 2002a; Schwarz et al., 2001), and the nutrient export associated with various land uses and pollutant sources (e.g., Alexander et al., 2001, 2004; Alexander, Elliott et al., 2002; 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.

B.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.

B.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.

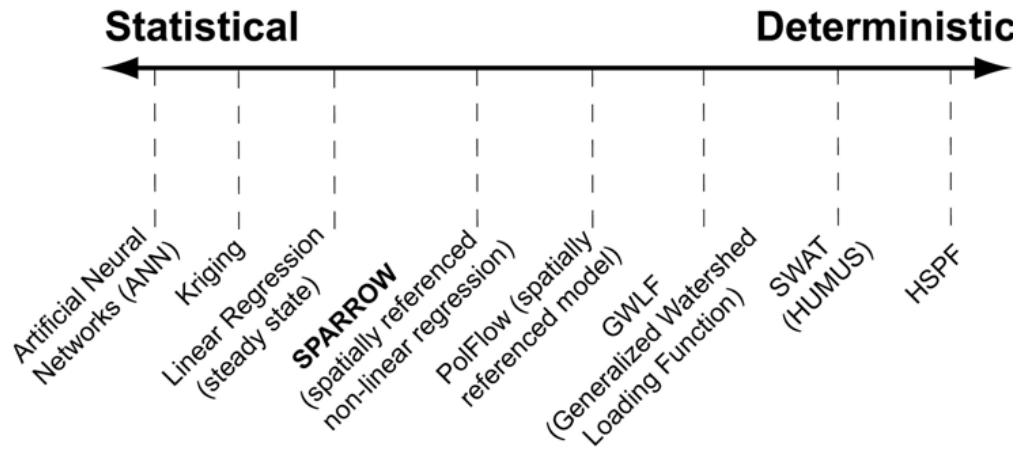
The spatial variability of a given variable is most readily increased in SPARROW models by expanding the spatial domain of the model to include larger drainage areas. This, of course, has the effect of increasing the number of monitoring stations, which also contributes to the potential for greater model 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;

Alexander et al., 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.

B.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 B-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 B-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., Caraco et al., 2003; Peierls et al., 1991; Howarth et al., 1996). 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., Hydrologic Simulation Program-Fortran (HSPF), Bicknell et al., 2001; Soil Water Assessment Tool (SWAT), Srinivasan et al., 1993; Agricultural Nonpoint Source Model (AGNPS), Young et al., 1995]. 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 (Beven, 2002; Jakeman and Hornberger, 1993). 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.

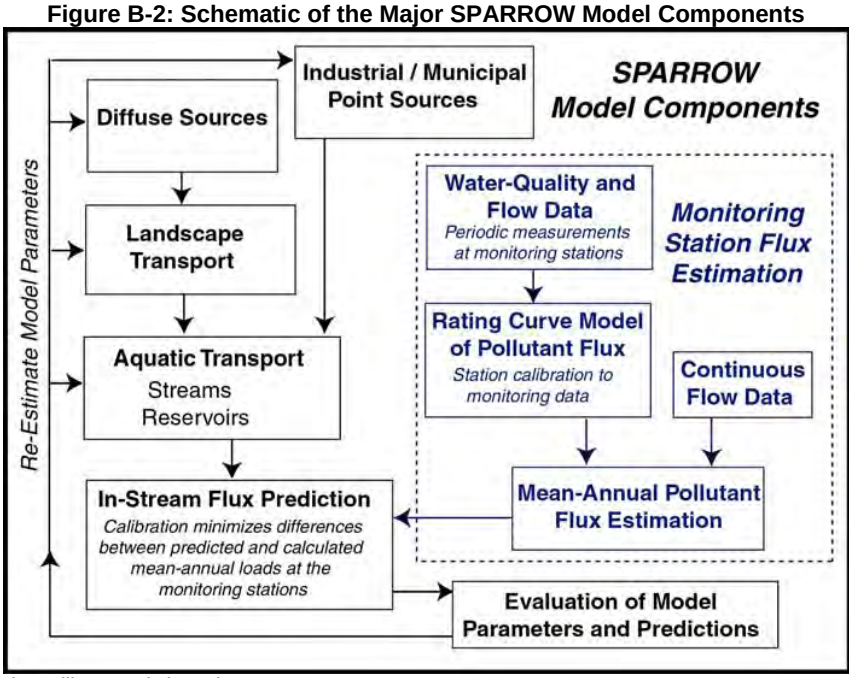
By comparison to other types of water-quality models, SPARROW may be best characterized as a hybrid process-based and statistical modeling approach for estimating pollutant sources and contaminant transport in surface waters. The mechanistic mass transport components of SPARROW include surface-water 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; Alexander et al., 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.

B.2.6 Model Infrastructure

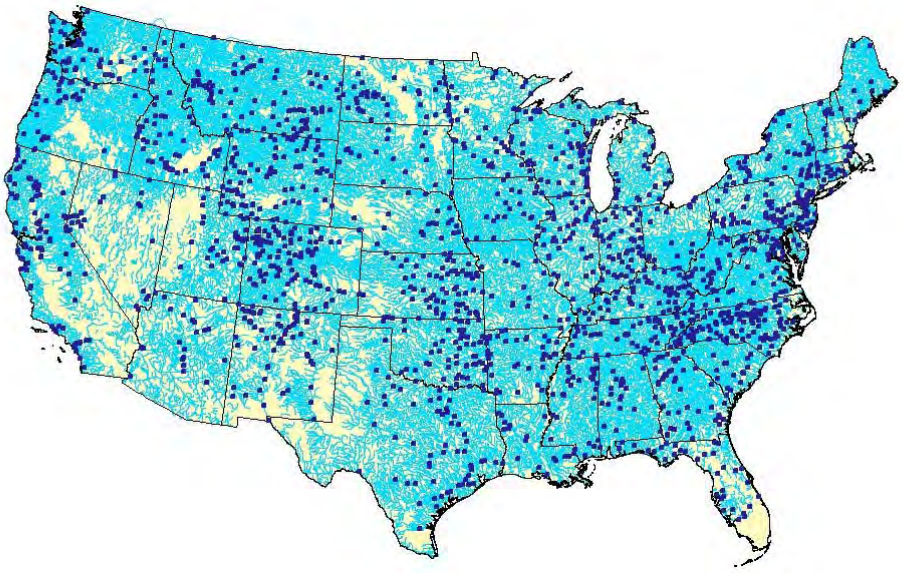
The parameters of SPARROW models are statistically estimated with nonlinear regression techniques by spatially correlating water-quality flux estimates at monitoring stations with watershed data on sources, and landscape and surface-water properties that affect transport. The calibrated models are then used to predict flux, total and disaggregated by contributing source, for stream reaches throughout a river network. A flow diagram is provided in *Figure B-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 B-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 B-2: Schematic of the Major SPARROW Model Components



Modified from Alexander, Elliott, et al. (2002).

Figure B-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



B.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.

B.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

$$\tilde{c}_i = M(Q_t)\beta_Q + h(T_t)\beta_T + X_t\beta_X + e_t \qquad \text{(B-1)}$$

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.

B.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 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.

B.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, Elliott, et al. (2002). 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

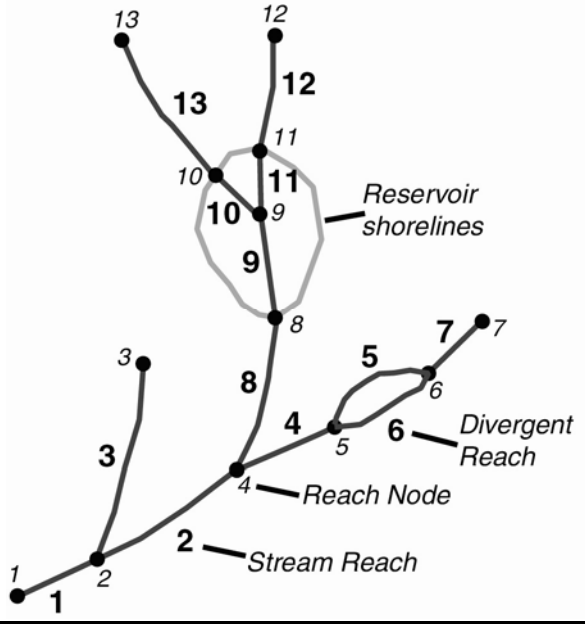
problems in assessing the error of SPARROW predictions if the predictions are conditioned on observed, upstream flux estimates. In national analyses, we have frequently excluded mean flux estimates that have a standard error greater than about 20 to 30 percent.

B.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 B-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 B-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 B-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 B-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 B-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 B-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 water body (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.

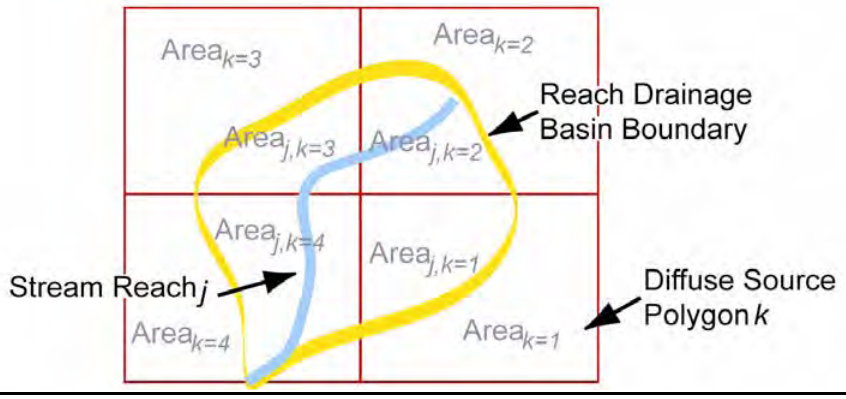
B.2.12 Watershed Sources and Explanatory Variables

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 B-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 B-5)*, estimates of the area-weighted sum of source characteristics ($S_{n,j}$) for reach *j* and source type *n* can be calculated as:

$$S_{n,j} = \sum_{k \in P(j)} S_{n,k} \left(A_{j,k} / A^{*}_{k} \right) \qquad \text{(B-2)}$$

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 B-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 B-2 be redefined to represent

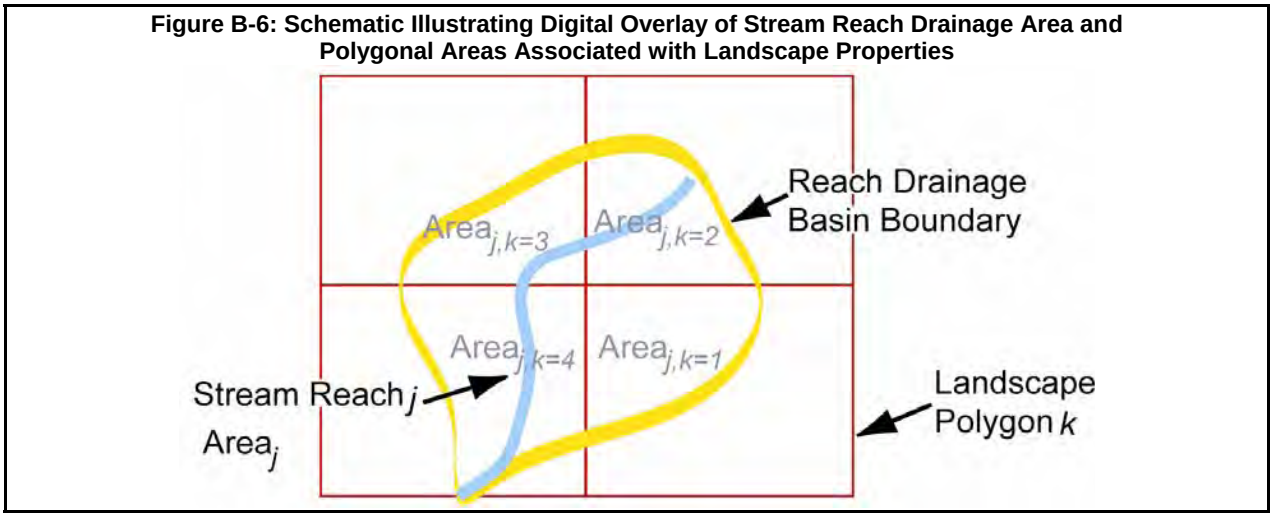
the area of the associated land use that intersects the reach *j* incremental drainage and source polygon *k*, and that the area term A^*_k be redefined to equal the total area of the associated land use within the source polygon *k*.

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:

$$Z_{i,j} = \sum_{k \in P(j)} \tilde{Z}_{i,k} \left(A_{j,k} / A^*_j \right) \qquad \text{(B-3)}$$

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 B-6* shows the relation between areas of landscape polygons and the incremental drainage basin of a hypothetical stream reach that is described in Equation B-3. Note that the area-weighted mean estimate defined in Equation B-3 differs from the area-weighted sum given by Equation B-2 in that the area ratio terms sum to 1.0 in Equation B-3 but not in Equation B-2.

Figure B-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.

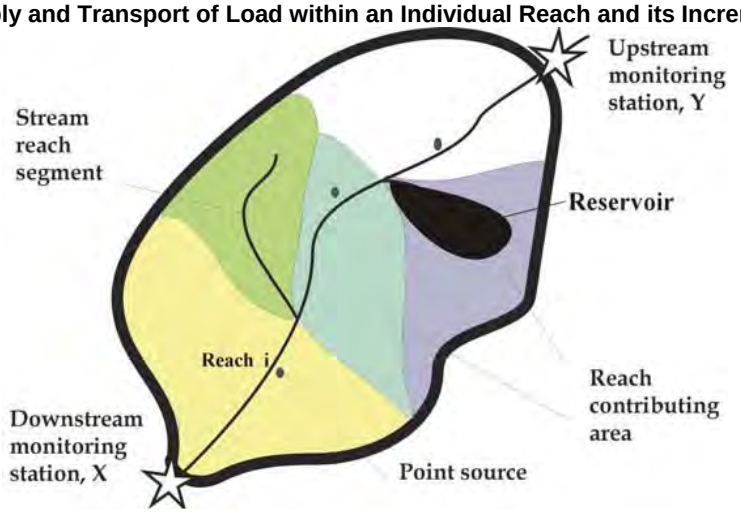
B.3 Model Specification

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.

B.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 B-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 B-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 B-7: Conceptual Illustration of a Reach Network for Five Incremental Watersheds. Model Equation B-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 B-7*). The functional relations determining reach *i* flux are given by:

$$F^*_i = \left[\sum_{j \in J(i)} F'_j\right] \delta_i A\left(Z^S{}_i, Z^R{}_i; \theta_S, \theta_R\right) + \left[\sum_{n=1}^{N_S} S_{n,i} \alpha_n D_n\left(Z^D{}_i; \theta_D\right)\right] A'\left(Z_i{}^S, Z_i{}^R; \theta_S, \theta_R\right) \qquad \text{(B-4)}$$

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 θ_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 B-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.

B.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 (Johnes et al., 1996; Beaulac and Reckhow, 1982). 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)

B.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 B-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

$$D_n\left(Z_i{}^D;\theta_D\right)=\exp\left\{\sum_{m=1}^{M_D}\omega_{nm}Z_{m,i}{}^D\theta_{Dm}\right\} \qquad \text{(B-5)}$$

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 B-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 B-5 be expressed as differences from their mean value over all reaches.

B.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 ($T_c{}^S{}_i$; units of time) in reach *i* and stream class *c* defined according to discrete intervals of mean streamflow or depth (in this case, $Z_i^S = \{T_c{}^S{}_i\}$, $c = 1,\ldots,C_S$, where $T_c{}^S{}_i$ 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:

$$A\left(Z_i{}^S, Z_i{}^R; \theta_S, \theta_R\right) = \exp\left\{-\sum_{c=1}^{C_S} \theta_{Sc} T_c{}^S{}_i\right\} \qquad \text{(B-6)}$$

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., USEPA, 1996b; Alexander et al., 1999; Jobson, 1996; Jowett, 1998) 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 B-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 McMahon et al., 2003; Alexander et al., 2002a).

B.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

$$A\left(\mathbf{Z}_i^S, \mathbf{Z}_i^R; \boldsymbol{\theta}_S, \boldsymbol{\theta}_R\right) = \frac{1}{1 + \theta_{R0}\left(q_i^R\right)^{-1}} \qquad \text{(B-7)}$$

where θ_{R0} is a parameter, to be estimated in the model, representing the apparent settling velocity.

B.4 Model Estimation

The SPARROW model equation, given in Equation B-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.

B.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.

B.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 B-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

$$F'_{j,n} = \frac{F^{*}_{j,n}}{\sum_{m=1}^{N_y} F^{*}_{j,m}} \cdot F_j^{M} \qquad \text{(B-8)}$$

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

$$F^{*}_{i,n} = \left[\sum_{j \in J(i)} F'_{j,n} \right] \delta_i A\left(Z^{S}{}_i, Z^{R}{}_i; \theta_S, \theta_R\right) + S_{n,i} \alpha_n D_n\left(Z_i^{D}; \theta_D\right) A'\left(Z_i^{S}, Z_i^{R}; \theta_S, \theta_R\right) \qquad \text{(B-9)}$$

Appendix C – A Preliminary SPARROW Model of Suspended Sediment for the Conterminous United States (Schwarz 2008a)

This appendix reprints in full the USGS Open File Report detailing the specific application of the SPARROW model to predict sediment concentrations in the RF1 stream network.

Appendix D – Water Quality Index Tables

This appendix presents detailed versions of *Table 10-5* through *Table 10-7*, showing WQI river mile and percentage improvements by baseline water quality index range, improvement range, and EPA region for each policy option.

Table D-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		
		RFI Miles Receiving C&D Discharges	Miles of River in RF1 Network	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles
1	WQI<26	30	30	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	26<WQI<50	916	1,130	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	50<WQI<70	12,343	12,558	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	70>WQI	3,663	4,575	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	Total	**16,952**	**18,293**	**0**	**0.0%**	**0.0%**	**0**	**0.0%**	**0.0%**	**0**	**0.0%**	**0.0%**	**0**	**0.0%**	**0.0%**
2	WQI<26	52	52	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	26<WQI<50	2,394	2,579	20	0.8%	0.8%	0	0.0%	0.0%	0	0.0%	0.0%	20	0.8%	0.8%
	50<WQI<70	4,824	4,846	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	70>WQI	8,617	8,633	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	Total	**15,887**	**16,110**	**20**	**0.1%**	**0.1%**	**0**	**0.0%**	**0.0%**	**0**	**0.0%**	**0.0%**	**20**	**0.1%**	**0.1%**
3	WQI<26	1,371	1,371	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	26<WQI<50	13,438	15,450	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	50<WQI<70	12,695	13,321	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	70>WQI	1,679	3,475	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	Total	**29,183**	**33,617**	**0**	**0.0%**	**0.0%**	**0**	**0.0%**	**0.0%**	**0**	**0.0%**	**0.0%**	**0**	**0.0%**	**0.0%**
4	WQI<26	1,053	1,065	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	26<WQI<50	37,102	37,828	1,793	4.8%	4.7%	356	1.0%	0.9%	123	0.3%	0.3%	2,272	6.1%	6.0%
	50<WQI<70	34,667	37,287	4,102	11.8%	11.0%	214	0.6%	0.6%	31	0.1%	0.1%	4,347	12.5%	11.7%
	70>WQI	14,490	18,284	2,313	16.0%	12.6%	168	1.2%	0.9%	232	1.6%	1.3%	2,712	18.7%	14.8%
	Total	**87,312**	**94,464**	**8,207**	**9.4%**	**8.7%**	**738**	**0.8%**	**0.8%**	**386**	**0.4%**	**0.4%**	**9,331**	**10.7%**	**9.9%**
5	WQI<26	6,402	6,402	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	26<WQI<50	44,892	44,929	46	0.1%	0.1%	0	0.0%	0.0%	0	0.0%	0.0%	46	0.1%	0.1%
	50<WQI<70	17,591	17,596	51	0.3%	0.3%	0	0.0%	0.0%	0	0.0%	0.0%	51	0.3%	0.3%
	70>WQI	2,624	2,624	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	Total	**71,509**	**71,550**	**96**	**0.1%**	**0.1%**	**0**	**0.0%**	**0.0%**	**0**	**0.0%**	**0.0%**	**96**	**0.1%**	**0.1%**

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		RFI Miles Receiving C&D Discharges	Miles of River in RF1 Network	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles
6	WQI<26	2,111	2,332	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	26<WQI<50	60,261	73,189	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	50<WQI<70	16,527	17,718	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	70>WQI	3,957	5,430	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	Total	**82,856**	**98,669**	**0**	**0.0%**	**0.0%**	**0**	**0.0%**	**0.0%**	**0**	**0.0%**	**0.0%**	**0**	**0.0%**	**0.0%**
7	WQI<26	15,139	16,103	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	26<WQI<50	35,697	41,624	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	50<WQI<70	2,965	2,965	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	70>WQI	217	217	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	Total	**54,018**	**60,909**	**0**	**0.0%**	**0.0%**	**0**	**0.0%**	**0.0%**	**0**	**0.0%**	**0.0%**	**0**	**0.0%**	**0.0%**
8	WQI<26	5,741	8,426	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	26<WQI<50	62,368	76,218	4	<0.0%	<0.0%	0	0.0%	0.0%	0	0.0%	0.0%	4	<0.0%	<0.0%
	50<WQI<70	28,974	29,040	28	0.1%	0.1%	0	0.0%	0.0%	0	0.0%	0.0%	28	0.1%	0.1%
	70>WQI	16,627	16,627	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	Total	**113,710**	**130,311**	**32**	**<0.0%**	**<0.0%**	**0**	**0.0%**	**0.0%**	**0**	**0.0%**	**0.0%**	**32**	**<0.0%**	**<0.0%**
9	WQI<26	9,345	13,040	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	26<WQI<50	35,035	38,566	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	50<WQI<70	2,833	3,468	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	70>WQI	261	1,185	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	Total	**47,474**	**56,259**	**0**	**0.0%**	**0.0%**	**0**	**0.0%**	**0.0%**	**0**	**0.0%**	**0.0%**	**0**	**0.0%**	**0.0%**
10	WQI<26	50	50	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	26<WQI<50	13,754	13,862	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	50<WQI<70	20,140	20,533	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	70>WQI	32,525	35,050	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	Total	**66,469**	**69,495**	**0**	**0.0%**	**0.0%**	**0**	**0.0%**	**0.0%**	**0**	**0.0%**	**0.0%**	**0**	**0.0%**	**0.0%**

Table D-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		
		RFI Miles Receiving C&D Discharges	Miles of River in RF1 Network	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles
National Total	WQI<26	41,294	48,871	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	26<WQI<50	305,857	345,376	1,862	0.6%	0.5%	356	0.1%	0.1%	123	0.0%	0.0%	2,341	0.8%	0.7%
	50<WQI<70	153,559	159,331	4,181	2.7%	2.6%	214	0.1%	0.1%	31	0.0%	0.0%	4,426	2.9%	2.8%
	70>WQI	84,660	96,099	2,313	2.7%	2.4%	168	0.2%	0.2%	232	0.3%	0.2%	2,712	3.2%	2.8%
	Total	**585,370**	**649,676**	**8,356**	**1.4%**	**1.3%**	**738**	**0.1%**	**0.1%**	**386**	**0.1%**	**0.1%**	**9,479**	**1.6%**	**1.5%**

Table D-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		
		RFI Miles Receiving C&D Discharges	Miles of River in RF1 Network	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles
1	WQI<26	30	30	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	26<WQI<50	916	1,130	59	6.4%	5.2%	0	0.0%	0.0%	0	0.0%	0.0%	59	6.4%	5.2%
	50<WQI<70	12,343	12,558	794	6.4%	6.3%	32	0.3%	0.3%	0	0.0%	0.0%	826	6.7%	6.6%
	70>WQI	3,663	4,575	658	18.0%	14.4%	6	0.2%	0.1%	0	0.0%	0.0%	664	18.1%	14.5%
	Total	**16,952**	**18,293**	**1,510**	**8.9%**	**8.3%**	**38**	**0.2%**	**0.2%**	**0**	**0.0%**	**0.0%**	**1,548**	**9.1%**	**8.5%**
2	WQI<26	52	52	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	26<WQI<50	2,394	2,579	698	29.2%	27.1%	30	1.2%	1.2%	7	0.3%	0.3%	735	30.7%	28.5%
	50<WQI<70	4,824	4,846	2,210	45.8%	45.6%	78	1.6%	1.6%	0	0.0%	0.0%	2,288	47.4%	47.2%
	70>WQI	8,617	8,633	2,437	28.3%	28.2%	88	1.0%	1.0%	0	0.0%	0.0%	2,525	29.3%	29.3%
	Total	**15,887**	**16,110**	**5,345**	**33.6%**	**33.2%**	**196**	**1.2%**	**1.2%**	**7**	**0.0%**	**0.0%**	**5,548**	**34.9%**	**34.4%**
3	WQI<26	1,371	1,371	283	20.6%	20.6%	65	4.8%	4.8%	11	0.8%	0.8%	359	26.2%	26.2%
	26<WQI<50	13,438	15,450	7,438	55.3%	48.1%	774	5.8%	5.0%	116	0.9%	0.8%	8,328	62.0%	53.9%
	50<WQI<70	12,695	13,321	7,517	59.2%	56.4%	1,390	11.0%	10.4%	117	0.9%	0.9%	9,024	71.1%	67.7%
	70>WQI	1,679	3,475	1,006	59.9%	29.0%	126	7.5%	3.6%	0	0.0%	0.0%	1,133	67.5%	32.6%
	Total	**29,183**	**33,617**	**16,244**	**55.7%**	**48.3%**	**2,356**	**8.1%**	**7.0%**	**244**	**0.8%**	**0.7%**	**18,844**	**64.6%**	**56.1%**
4	WQI<26	1,053	1,065	57	5.4%	5.3%	0	0.0%	0.0%	0	0.0%	0.0%	57	5.4%	5.3%
	26<WQI<50	37,102	37,828	8,247	22.2%	21.8%	2,840	7.7%	7.5%	1,506	4.1%	4.0%	12,593	33.9%	33.3%
	50<WQI<70	34,667	37,287	15,799	45.6%	42.4%	7,201	20.8%	19.3%	2,412	7.0%	6.5%	25,412	73.3%	68.2%
	70>WQI	14,490	18,284	6,066	41.9%	33.2%	3,202	22.1%	17.5%	899	6.2%	4.9%	10,168	70.2%	55.6%
	Total	**87,312**	**94,464**	**30,169**	**34.6%**	**31.9%**	**13,243**	**15.2%**	**14.0%**	**4,817**	**5.5%**	**5.1%**	**48,229**	**55.2%**	**51.1%**

Table D-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		
		RFI Miles Receiving C&D Discharges	Miles of River in RF1 Network	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles
5	WQI<26	6,402	6,402	8	0.1%	0.1%	0	0.0%	0.0%	0	0.0%	0.0%	8	0.1%	0.1%
	26<WQI<50	44,892	44,929	3,328	7.4%	7.4%	158	0.4%	0.4%	57	0.1%	0.1%	3,542	7.9%	7.9%
	50<WQI<70	17,591	17,596	3,326	18.9%	18.9%	743	4.2%	4.2%	81	0.5%	0.5%	4,150	23.6%	23.6%
	70>WQI	2,624	2,624	494	18.8%	18.8%	89	3.4%	3.4%	0	0.0%	0.0%	582	22.2%	22.2%
	Total	**71,509**	**71,550**	**7,156**	**10.0%**	**10.0%**	**990**	**1.4%**	**1.4%**	**138**	**0.2%**	**0.2%**	**8,283**	**11.6%**	**11.6%**
6	WQI<26	2,111	2,332	89	4.2%	3.8%	0	0.0%	0.0%	27	1.3%	1.2%	116	5.5%	5.0%
	26<WQI<50	60,261	73,189	1,978	3.3%	2.7%	1,107	1.8%	1.5%	733	1.2%	1.0%	3,818	6.3%	5.2%
	50<WQI<70	16,527	17,718	3,067	18.6%	17.3%	4,476	27.1%	25.3%	2,694	16.3%	15.2%	10,237	61.9%	57.8%
	70>WQI	3,957	5,430	599	15.1%	11.0%	1,267	32.0%	23.3%	471	11.9%	8.7%	2,337	59.0%	43.0%
	Total	**82,856**	**98,669**	**5,734**	**6.9%**	**5.8%**	**6,850**	**8.3%**	**6.9%**	**3,924**	**4.7%**	**4.0%**	**16,507**	**19.9%**	**16.7%**
7	WQI<26	15,139	16,103	145	1.0%	0.9%	0	0.0%	0.0%	36	0.2%	0.2%	180	1.2%	1.1%
	26<WQI<50	35,697	41,624	1,636	4.6%	3.9%	187	0.5%	0.4%	157	0.4%	0.4%	1,980	5.5%	4.8%
	50<WQI<70	2,965	2,965	500	16.9%	16.9%	791	26.7%	26.7%	285	9.6%	9.6%	1,576	53.1%	53.2%
	70>WQI	217	217	29	13.5%	13.4%	26	12.2%	12.1%	0	0.0%	0.0%	56	25.6%	25.6%
	Total	**54,018**	**60,909**	**2,310**	**4.3%**	**3.8%**	**1,004**	**1.9%**	**1.6%**	**477**	**0.9%**	**0.8%**	**3,791**	**7.0%**	**6.2%**
8	WQI<26	5,741	8,426	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	26<WQI<50	62,368	76,218	32	0.1%	0.0%	0	0.0%	0.0%	9	0.0%	0.0%	41	0.1%	0.1%
	50<WQI<70	28,974	29,040	140	0.5%	0.5%	0	0.0%	0.0%	0	0.0%	0.0%	140	0.5%	0.5%
	70>WQI	16,627	16,627	107	0.6%	0.6%	0	0.0%	0.0%	0	0.0%	0.0%	107	0.6%	0.6%
	Total	**113,710**	**130,311**	**278**	**0.2%**	**0.2%**	**0**	**0.0%**	**0.0%**	**9**	**0.0%**	**0.0%**	**288**	**0.3%**	**0.2%**

Table D-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		
		RFI Miles Receiving C&D Discharges	Miles of River in RF1 Network	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles
9	WQI<26	9,345	13,040	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	26<WQI<50	35,035	38,566	331	0.9%	0.9%	40	0.1%	0.1%	0	0.0%	0.0%	371	1.1%	1.0%
	50<WQI<70	2,833	3,468	10	0.4%	0.3%	0	0.0%	0.0%	0	0.0%	0.0%	10	0.4%	0.3%
	70>WQI	261	1,185	0	0.0%	0.0%	6	2.4%	0.5%	0	0.0%	0.0%	6	2.4%	0.5%
	Total	**47,474**	**56,259**	**342**	**0.7%**	**0.6%**	**46**	**0.1%**	**0.1%**	**0**	**0.0%**	**0.0%**	**388**	**0.8%**	**0.7%**
10	WQI<26	50	50	167	334.0%	335.0%	0	0.0%	0.0%	0	0.0%	0.0%	167	334.0%	335.0%
	26<WQI<50	13,754	13,862	1,514	11.0%	10.9%	0	0.0%	0.0%	0	0.0%	0.0%	1,514	11.0%	10.9%
	50<WQI<70	20,140	20,533	2,864	14.2%	13.9%	0	0.0%	0.0%	0	0.0%	0.0%	2,864	14.2%	13.9%
	70>WQI	32,525	35,050	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	Total	**66,469**	**69,495**	**4,545**	**6.8%**	**6.5%**	**0**	**0.0%**	**0.0%**	**0**	**0.0%**	**0.0%**	**4,545**	**6.8%**	**6.5%**
National Total	WQI<26	41,294	48,871	748	1.8%	1.5%	65	0.2%	0.1%	74	0.2%	0.2%	888	2.1%	1.8%
	26<WQI<50	305,857	345,376	25,260	8.3%	7.3%	5,135	1.7%	1.5%	2,584	0.8%	0.7%	32,979	10.8%	9.5%
	50<WQI<70	153,559	159,331	36,228	23.6%	22.7%	14,711	9.6%	9.2%	5,589	3.6%	3.5%	56,528	36.8%	35.5%
	70>WQI	84,660	96,099	11,396	13.5%	11.9%	4,811	5.7%	5.0%	1,370	1.6%	1.4%	17,578	20.8%	18.3%
	Total	**585,370**	**649,676**	**73,632**	**12.6%**	**11.3%**	**24,723**	**4.2%**	**3.8%**	**9,616**	**1.6%**	**1.5%**	**107,972**	**18.4%**	**16.6%**

Table D-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		
		RFI Miles Receiving C&D Discharges	Miles of River in RF1 Network	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles
1	WQI<26	30	30	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	26<WQI<50	916	1,130	210	23.0%	18.6%	0	0.0%	0.0%	0	0.0%	0.0%	210	23.0%	18.6%
	50<WQI<70	12,343	12,558	1,627	13.2%	13.0%	112	0.9%	0.9%	9	0.1%	0.1%	1,748	14.2%	13.9%
	70>WQI	3,663	4,575	887	24.2%	19.4%	81	2.2%	1.8%	0	0.0%	0.0%	968	26.4%	21.2%
	Total	**16,952**	**18,293**	**2,725**	**16.1%**	**14.9%**	**192**	**1.1%**	**1.1%**	**9**	**0.1%**	**0.0%**	**2,926**	**17.3%**	**16.0%**
2	WQI<26	52	52	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	26<WQI<50	2,394	2,579	1,013	42.3%	39.3%	107	4.5%	4.1%	21	0.9%	0.8%	1,140	47.6%	44.2%
	50<WQI<70	4,824	4,846	2,684	55.6%	55.4%	307	6.4%	6.3%	9	0.2%	0.2%	3,001	62.2%	61.9%
	70>WQI	8,617	8,633	3,167	36.8%	36.7%	207	2.4%	2.4%	0	0.0%	0.0%	3,373	39.1%	39.1%
	Total	**15,887**	**16,110**	**6,864**	**43.2%**	**42.6%**	**621**	**3.9%**	**3.9%**	**30**	**0.2%**	**0.2%**	**7,514**	**47.3%**	**46.6%**
3	WQI<26	1,371	1,371	255	18.6%	18.6%	93	6.8%	6.8%	20	1.5%	1.5%	369	26.9%	26.9%
	26<WQI<50	13,438	15,450	7,951	59.2%	51.5%	1,358	10.1%	8.8%	186	1.4%	1.2%	9,495	70.7%	61.5%
	50<WQI<70	12,695	13,321	7,566	59.6%	56.8%	1,980	15.6%	14.9%	142	1.1%	1.1%	9,688	76.3%	72.7%
	70>WQI	1,679	3,475	1,071	63.8%	30.8%	199	11.9%	5.7%	7	0.4%	0.2%	1,278	76.1%	36.8%
	Total	**29,183**	**33,617**	**16,843**	**57.7%**	**50.1%**	**3,631**	**12.4%**	**10.8%**	**355**	**1.2%**	**1.1%**	**20,830**	**71.4%**	**62.0%**
4	WQI<26	1,053	1,065	55	5.2%	5.1%	31	2.9%	2.9%	6	0.6%	0.6%	91	8.7%	8.6%
	26<WQI<50	37,102	37,828	8,192	22.1%	21.7%	4,318	11.6%	11.4%	2,665	7.2%	7.0%	15,175	40.9%	40.1%
	50<WQI<70	34,667	37,287	13,599	39.2%	36.5%	10,446	30.1%	28.0%	4,110	11.9%	11.0%	28,154	81.2%	75.5%
	70>WQI	14,490	18,284	4,999	34.5%	27.3%	4,603	31.8%	25.2%	1,705	11.8%	9.3%	11,307	78.0%	61.8%
	Total	**87,312**	**94,464**	**26,844**	**30.7%**	**28.4%**	**19,397**	**22.2%**	**20.5%**	**8,486**	**9.7%**	**9.0%**	**54,727**	**62.7%**	**57.9%**
5	WQI<26	6,402	6,402	222	3.5%	3.5%	0	0.0%	0.0%	0	0.0%	0.0%	222	3.5%	3.5%
	26<WQI<50	44,892	44,929	4,880	10.9%	10.9%	458	1.0%	1.0%	82	0.2%	0.2%	5,420	12.1%	12.1%
	50<WQI<70	17,591	17,596	3,657	20.8%	20.8%	1,214	6.9%	6.9%	332	1.9%	1.9%	5,203	29.6%	29.6%
	70>WQI	2,624	2,624	551	21.0%	21.0%	210	8.0%	8.0%	0	0.0%	0.0%	761	29.0%	29.0%
	Total	**71,509**	**71,550**	**9,310**	**13.0%**	**13.0%**	**1,883**	**2.6%**	**2.6%**	**414**	**0.6%**	**0.6%**	**11,606**	**16.2%**	**16.2%**

Table D-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		
		RFI Miles Receiving C&D Discharges	Miles of River in RF1 Network	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles
6	WQI<26	2,111	2,332	92	4.3%	3.9%	34	1.6%	1.5%	62	3.0%	2.7%	188	8.9%	8.1%
	26<WQI<50	60,261	73,189	1,775	2.9%	2.4%	1,349	2.2%	1.8%	1,148	1.9%	1.6%	4,273	7.1%	5.8%
	50<WQI<70	16,527	17,718	2,829	17.1%	16.0%	4,423	26.8%	25.0%	3,560	21.5%	20.1%	10,811	65.4%	61.0%
	70>WQI	3,957	5,430	517	13.1%	9.5%	1,288	32.6%	23.7%	625	15.8%	11.5%	2,430	61.4%	44.8%
	Total	**82,856**	**98,669**	**5,212**	**6.3%**	**5.3%**	**7,095**	**8.6%**	**7.2%**	**5,395**	**6.5%**	**5.5%**	**17,702**	**21.4%**	**17.9%**
7	WQI<26	15,139	16,103	207	1.4%	1.3%	19	0.1%	0.1%	56	0.4%	0.3%	281	1.9%	1.7%
	26<WQI<50	35,697	41,624	2,138	6.0%	5.1%	209	0.6%	0.5%	256	0.7%	0.6%	2,603	7.3%	6.3%
	50<WQI<70	2,965	2,965	599	20.2%	20.2%	788	26.6%	26.6%	423	14.3%	14.3%	1,810	61.0%	61.0%
	70>WQI	217	217	29	13.5%	13.4%	26	12.2%	12.1%	0	0.0%	0.0%	56	25.6%	25.6%
	Total	**54,018**	**60,909**	**2,973**	**5.5%**	**4.9%**	**1,041**	**1.9%**	**1.7%**	**735**	**1.4%**	**1.2%**	**4,749**	**8.8%**	**7.8%**
8	WQI<26	5,741	8,426	22	0.4%	0.3%	0	0.0%	0.0%	0	0.0%	0.0%	22	0.4%	0.3%
	26<WQI<50	62,368	76,218	520	0.8%	0.7%	96	0.2%	0.1%	57	0.1%	0.1%	673	1.1%	0.9%
	50<WQI<70	28,974	29,040	2,116	7.3%	7.3%	127	0.4%	0.4%	11	0.0%	0.0%	2,254	7.8%	7.8%
	70>WQI	16,627	16,627	945	5.7%	5.7%	138	0.8%	0.8%	60	0.4%	0.4%	1,142	6.9%	6.9%
	Total	**113,710**	**130,311**	**3,603**	**3.2%**	**2.8%**	**362**	**0.3%**	**0.3%**	**128**	**0.1%**	**0.1%**	**4,092**	**3.6%**	**3.1%**
9	WQI<26	9,345	13,040	138	1.5%	1.1%	0	0.0%	0.0%	0	0.0%	0.0%	138	1.5%	1.1%
	26<WQI<50	35,035	38,566	1,518	4.3%	3.9%	306	0.9%	0.8%	95	0.3%	0.2%	1,919	5.5%	5.0%
	50<WQI<70	2,833	3,468	130	4.6%	3.7%	0	0.0%	0.0%	0	0.0%	0.0%	130	4.6%	3.7%
	70>WQI	261	1,185	0	0.0%	0.0%	0	0.0%	0.0%	6	2.4%	0.5%	6	2.4%	0.5%
	Total	**47,474**	**56,259**	**1,786**	**3.8%**	**3.2%**	**306**	**0.6%**	**0.5%**	**101**	**0.2%**	**0.2%**	**2,193**	**4.6%**	**3.9%**
10	WQI<26	50	50	0	0.0%	0.0%		0.0%	0.0%	0	0.0%	0.0%	0	0.0%	0.0%
	26<WQI<50	13,754	13,862	282	2.1%	2.0%	133	1.0%	1.0%	68	0.5%	0.5%	483	3.5%	3.5%
	50<WQI<70	20,140	20,533	1,623	8.1%	7.9%	886	4.4%	4.3%	308	1.5%	1.5%	2,818	14.0%	13.7%
	70>WQI	32,525	35,050	2,584	7.9%	7.4%	2,349	7.2%	6.7%	1,886	5.8%	5.4%	6,818	21.0%	19.5%
	Total	**66,469**	**69,495**	**4,489**	**6.8%**	**6.5%**	**3,367**	**5.1%**	**4.8%**	**2,261**	**3.4%**	**3.3%**	**10,118**	**15.2%**	**14.6%**

Table D-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		
		RFI Miles Receiving C&D Discharges	Miles of River in RF1 Network	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles	River Miles	% of River Miles Receiving C&D Discharges	% of Total River Miles
National Total	WQI<26	41,294	48,871	990	2.4%	2.0%	177	0.4%	0.4%	144	0.3%	0.3%	1,311	3.2%	2.7%
	26<WQI<50	305,857	345,376	28,480	9.3%	8.2%	8,333	2.7%	2.4%	4,578	1.5%	1.3%	41,391	13.5%	12.0%
	50<WQI<70	153,559	159,331	36,429	23.7%	22.9%	20,284	13.2%	12.7%	8,904	5.8%	5.6%	65,617	42.7%	41.2%
	70>WQI	84,660	96,099	14,750	17.4%	15.3%	9,102	10.8%	9.5%	4,288	5.1%	4.5%	28,139	33.2%	29.3%
	Total	**585,370**	**649,676**	**80,648**	**13.8%**	**12.4%**	**37,895**	**6.5%**	**5.8%**	**17,915**	**3.1%**	**2.8%**	**136,458**	**23.3%**	**21.0%**

Appendix E – 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 proposed C&D 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 (Bergstrom and Taylor 2006; Johnston et al. 2005; 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 2004c); 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 water bodies 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.

E.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 1985 and 2004, that applied generally accepted, standard valuation methods to determine total (including use and non-use) values

associated with aquatic habitat improvements.[18] 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. 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 to 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 surplus 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. Results of preliminary models estimated by EPA support this conclusion; preliminary models that attempted to combine both stated and revealed preference studies generated poorly fitting, counter-intuitive, and often nonsensical results. For this reason, all final models impose welfare consistency and limit analyses to metadata including only stated preference studies.

E.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:

[18] Data were collected from 66 studies. Nine studies using revealed preference techniques were dropped from the model since combination of stated and revealed preference studies violates the requirement of welfare consistency. Additional studies were dropped from the analysis because they utilized nonstandard methodologies or did not provide information for key regressors.

- Review of EPA's research and bibliographies dealing with non-market benefits associated with water quality changes
- Systematic review of recent issues of 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 contingent valuation studies and or water quality 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), Natural Resource Conservation Service (NRCS), National Bureau of Economic Research (NBER)).

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-six of these studies met the criteria identified for inclusion in the meta-analysis.[19] 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 water bodies, including aquatic life support, recreational activities (such as fishing, boating, and swimming), and non-use value
- **Values estimated:** Selected studies were limited to those that elicited total household WTP (the total of use and non-use components)
- **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 66 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 (66) 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

[19] 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).

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).* 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 proposed C&D regulation (DCN 6-1900), 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 water body 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)[20] 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).

E.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 2002, 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 and Canada. 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

[20] Additional details on the WQI and the use of the WQI in survey instruments are provided by McClelland (1974), Mitchell and Carson (1989, p. 342), and Vaughan (1986). 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 water body.

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), water body type, number of water bodies affected, recreational activities affected by the quality change, and species affected by the quality change. *Table E-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; 10 studies collected information through personal interviews in the home, onsite, or in a centralized location, and 9 surveys were conducted by telephone. Study sample sizes range from 96 to 2,907 responses.

Table E-1: Selected Summary Information for Studies

Author(s) and Year	Observations	State	Water Body 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. (1989a)	1	DC, MD, VA	estuary	general water quality; phosphorus and nitrogen	swimming, beach, boating, fishing, outings
Bockstael et al. (1989b)	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. (1987)	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
Magat et al. (2000)	4	CO, NC	all freshwater	general water quality	fishing and swimming; fishing

Table E-1: Selected Summary Information for Studies

Author(s) and Year	Observations	State	Water Body Type	Type of Water Quality Improvement	Affected Recreational Uses
Matthews et al. (1999)	2	MN	river/stream	phosphorus	boating and fishing
Opaluch et al. (1998)	1	NY	estuary	general water quality	shellfishing
Olsen et al. (1991)	3	ID, MT, OR, WA	river/stream	wildlife habitat	fishing
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
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, 28 presented only WTP values for changes in general water quality (approximately 62 percent). Of the studies that did address specific changes, nine specified nutrients and/or sediment, four addressed hazardous pollutants including heavy metals and pesticides, four addressed wildlife habitat, one addressed acid deposition, and one addressed fecal coliform bacteria. 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.

E.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.[21] 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 illustrative 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 (Shrestha et al. 2007; Rosenberger and Phipps 2007; USEPA 2000b 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 (Rosenberger and Phipps 2007; Johnston et al. 2005). 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., Bergstrom and Taylor 2006; Moeltner et al. 2007; Johnston et al. 2005; Rosenberger and Johnston 2007; Shrestha et al. 2007; Rosenberger and Phipps 2007). USEPA (2000b) characterizes meta-analysis as "the most rigorous" benefit transfer method. In contrast, the USEPA Science Advisory Board's Advisory on EPA's Issues in Valuing Mortality Risk Reduction (2007) 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

[21] In its analysis of non-use benefits for the final 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 the final Phase II rule for further details regarding both the log-log and semi-log regression models estimated in EPA's analysis of non-use benefits for the final Phase II rule (USEPA, 2004c; 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 et al. 2002; Smith and Pattanayak 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. 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.

E.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), Moeltner et al. (2007), Bergstrom and Taylor (2006), Johnston et al. (2005, 2006), 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). [22]

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 to

[22] 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 12E.2.2: Total WTP Regression Model and Results* for further details concerning the specification of the model).

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 $8.35 to $503.36, with a mean value of $115.37.

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 water bodies); 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, non-users); 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 non-users.

Geographic region and scale variables characterize such features as:

- The number of water bodies affected by the policy

- Whether the study considered water quality improvements in all water bodies 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.[23] 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 hand, and assumed population increases to correspond to modest increases in water quality in order to be

[23] 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.

conservative.[24] 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 E-2*.

Table E-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: 1.98 to 5.93)	4.45 (0.78)
year_indx	Year in which the study was conducted, converted to an index by subtracting 1980.	Year index (Range: 11 to 22)	9.49 (6.04)
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 (0.50)
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: 0.25 to 5.75)	2.17 (1.08)
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)	2.90 (0.60)
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)	56,806.24 (13,903.11)
Nonusers	Binary (dummy) variable indicating that the survey was implemented over a population of non-users (default category for this dummy is a survey of any population that includes users).	Binary (Range: 0 or 1)	0.09 (0.29)

[24] 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 E-2: Variables and Descriptive Statistics for the Total WTP Regression Model

Variable	Description	Units and Measurement	Mean (Std. Dev.)
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)	0.22 (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 water bodies. All others specified either 1 water body, or the number was unspecified.)	Number of specified rivers or ponds (Range: 0 to 15)	1.00 (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)	0.03 (0.18)
Allmult	Binary (dummy) variable indicating that either all or multiple aquatic species are affected by the resource change.	Binary (Range: 0 or 1)	0.20 (0.40)
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)	0.40 (0.49)
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: 5 to 70)	40.33 (20.07)
Lnbase	Natural log of baseline water quality, specified on the WQI.	WQI units (Range: 5 to 70)	3.48 (0.80)

Source: U.S. EPA analysis.

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.

E.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 E-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 (Bateman and Jones 2003; Poe et al. 2001; Rosenberger and Loomis 2000a, 2000b), 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 WSUT 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 ($p = 0.44$).

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 E-3 presents results of the total WTP regression model.

E.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 E-3)* show that model variables are jointly significant at $p<0.0001$ in both cases. In both models, the majority of independent variables are statistically significant at $p<0.05$, with most statistically significant at $p<0.01$. Signs of significant parameter estimates generally correspond with intuition, where prior expectations exist. As shown in *Table E-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.

Table E-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	5.5802[3]	1.0452	5.9892[3]	0.5566
year_indx	-0.07741[3]	0.02074	-0.05899[3]	0.01755
Discrete	-0.1198	0.225	-0.153	0.1673
Volunt	-1.3255[3]	0.1611	-1.2937[3]	0.155
Nonparam	-0.6732[3]	0.1386	-0.6919[3]	0.1122
Income	2.47×10^{-6}	3.9×10^{-6}	-6.10×10^{-7}	3.35×10^{-6}
WQI	-0.3238	0.2693	-0.3709[1]	0.2076
outlier_bids	-0.8736[3]	0.3028	-0.7108[3]	0.2133
single_river	-0.3971[3]	0.1286	-0.225[1]	0.1162
single_lake	-0.0575	0.2333	-0.1391	0.2161
multiple_river	-1.4554[3]	0.3768	-1.4485[3]	0.3086
regional_fresh	0.1773	0.1587	0.2698[2]	0.129
salt_pond	1.0251[3]	0.3718	1.2163[3]	0.3181
num_riv_pond	0.1155[3]	0.02859	0.1104[3]	0.02346
Mr	-0.8784[3]	0.1783	-0.8014[3]	0.1226
Mp	1.5114[3]	0.2968	1.3534[3]	0.2042
Nonusers	-0.4093[3]	0.1303	-0.4512[3]	0.1278
Allmult	-0.3755[2]	0.1597	-0.4073[3]	0.1189
Nonspec	-0.3849[1]	0.2012	-0.2419	0.155
Quality_ch (lnquality_ch in trans-log model)	0.4359[3]	0.1471	0.03405[3]	0.006313
fish_use	-0.3478[3]	0.1165	-0.4505[3]	0.08377
Fishplus	0.4343[2]	0.1671	0.3781[3]	0.1108
baseline (*lnbase* in trans-log model)	0.0317	0.1177	0.00577	0.003655
Mail	-0.2198	0.1559	-0.345[3]	0.1199

Table E-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
lump_sum	0.5468[2]	0.2364	0.7039[3]	0.1728
non_reviewed	-0.2757[1]	0.1653	-0.4007[3]	0.1205
Median_WTP	-0.514[3]	0.188	-0.4399[2]	0.1986
-2 Log Likelihood	134.3		115.4	
Covariance Factors:				
Study Level (σ_u)	0		0	
Residual (σ_e)	0.1883		0.1597	

Source: USEPA analysis

1 $p > |t| < 0.1$
2 $p > |t| < 0.05$
3 $p > |t| < 0.01$

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 $p<0.0001$. 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 $p<0.12$ 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 ($p<0.01$) 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 ($p<0.01$) 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 water body. 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 water bodies* under consideration and geographic scale of improvement (*regional_fresh*). Of the water body categories distinguished above, both rivers and salt ponds allowed variation in numbers of affected water bodies explicitly described by the survey. This variation is captured by the variable *num_riv_pond* (see *Table E-3)*.[25] The associated parameter estimate

[25] Technically, this variable is the sum of two interaction variables: (1) an interaction between multiple_river and the number of water bodies noted in the survey (0 if unspecified) and (2) an interaction between salt_pond and the number of water bodies noted in the survey (0 if unspecified).

is statistically significant ($p<0.01$) and indicates that WTP increases with the number of water bodies considered. The parameter estimate on the r*egional_fresh* variable is positive and significant ($p<0.05$) 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 scope, both in terms of the number of water bodies 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 ($p>0.26$).

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 ($p<.0001$) parameter estimate indicates that surveys of non-users only, who by definition have only non-use 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 non-use values. Based on this statistically significant result, it is possible to use this model to estimate non-use values, interpreted as the mean WTP values estimated by surveys of non-users only. Such methods, however, may underestimate non-use values of the general population, if the non-use values of users exceed those of non-users (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<0.05*) 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 ($p>0.16$) 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.

E.2.4 Model Selection

To select the model for estimating benefits of water quality improvements from the C&D 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 10-8* in *Chapter 10* provides a complete list of values assigned to the remaining independent regressors.

As shown in *Table E-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 C&D 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 E-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	0.1 Point Change in WQI
Semi-log	20.6584129	14.6967	12.39599	10.81758	10.4911
Trans-log	22.8032699	16.8814684	12.50388	6.247071	2.349331

Semi-log Error Term = 0.07985
Trans-log Error Term = 0.09415

E.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 non-use 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 (i.e., nutrients) that are different from the pollutant of concern in the C&D 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, 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, 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. Bergstrom and Taylor 2006; Shrestha et al. 2007; USEPA 2000b).

Based on the results presented in *Table E-3,* EPA estimates WTP for water resource changes as a function of resource, regional, and study design attributes (see *Chapter 10*). This, in general, provides a superior alternative to the calculation and use of a simple arithmetic mean over the 115 observations, as it allows WTP to be adjusted to account for the characteristics of the transfer site. The ability of the model to appropriately adjust WTP is suggested by the many systematic (statistically significant) patterns revealed by the meta-analysis regression model. Nonetheless, the use (and interpretation) of such WTP estimates for benefit transfer is subject to the constraints and concerns expressed elsewhere in the literature (e.g., Rosenberger and Johnston 2007; Desvousges et al. 1998; Poe et al. 2001; Vandenberg et al. 2001; Navrud and Ready 2007).